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The environmental sustainability of premium Australian beef exported to the USA: A Life Cycle Assessment

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Abstract

This study completed a Life cycle assessment (LCA) investigating resource use and environmental impacts from grain and grass finished beef production in two regions of eastern Australia. The system extended from production through to the wholesale distribution of a retail ready product in the USA. The study investigated energy demand, water use, land occupation, greenhouse gas emissions and stress weighted water use. Fossil fuel energy demand ranged from 24.2 to 44.3 MJ / kg boneless beef. Consumptive water use ranged from 410.2 to 640.3 L / kg boneless beef, while stress weighted water use ranged from 27.7 to 192.7 L H_2O-e / kg boneless beef. Occupation of cultivated land ranged from 0.0 to 28.7 m^2 / kg boneless beef. Human edible protein efficiency (higher than 1.0 meaning higher efficiency) was very high for grass finishing systems (11.0 to 107.8 in Queensland) but lower in the more intensive NSW grass finishing systems (1.8 to 12.3). Human edible protein efficiency was much lower for grain finishing systems, at 0.8 for the mid fed (115 days) and 0.4 for the long fed (330 days) scenarios. Greenhouse gas (GHG) emissions ranged from 19.8 to 27.1 kg CO_2-e / kg boneless beef, and increased by 6-34% when Land Use and direct Land Use Change sources were included. The production system contributed the majority of impacts across all categories, with meat processing, transport and storage collectively contributing <4% to GHG emissions. These results suggest that the impacts from the transport of beef, even where the distances are large, are not a significant driver of the environmental sustainability of this product.

Executive summary

Australia is the second largest global exporter of beef in the world after Brazil. The Australian beef industry maintains a strong emphasis on producing beef from sustainable production systems, predominantly from the extensive rangeland areas of eastern Australia. While a number of studies have been conducted to quantify the resource use and environmental impacts of Australian beef production, none have investigated the impacts of producing and exporting premium beef to the USA. Australia supplies both premium grass-fed and small volumes of high quality grain-finished beef to the USA, much of which is destined for the food services sector. This study investigated energy demand, consumptive water use, land occupation, greenhouse gas emissions and stress weighted water use associated with producing, processing and exporting Australian beef from two major production regions in eastern Australia to the USA. The study followed a Life Cycle Assessment (LCA) method for the production of one kilogram of retail ready, boneless beef in the USA. The specified end point was the distribution warehouse in the United States following production, processing, transport and storage.

Fossil fuel energy demand ranged from 24.2 to 44.3 MJ / kg boneless beef. Consumptive water use ranged from 410.2 to 640.3 L / kg boneless beef, while stress weighted water use ranged from 27.7 to 192.7 L H_2O-e / kg boneless beef. Occupation of cultivated land ranged from 0.0 to 28.7 m^2 / kg boneless beef, with most of the land used by Australian beef cattle being non-arable rangeland unsuitable for alternative food production systems. Human edible protein efficiency (HEPE) was included as a measure of the efficiency with which beef cattle convert human edible food sources such as grain into meat products. This measure excludes non-human edible food sources such as grass, and results above 1 indicate more meat protein is produced than consumed. Results showed that in the extensive northern region of Queensland, HEPE ranged from 11.0 to 107.8 and ranged from 1.8 to 12.3 for the NSW region. Values were lower from grain finished beef, ranging from 0.8 for the mid fed (115 days) to 0.4 for the long fed (330 days) production systems. Total greenhouse gas (GHG) emissions (excluding land use change) ranged from 19.8 to 27.1 kg CO_2-e / kg boneless beef. Mean GHG emissions from land use and direct land use change (LU and dLUC) over the 1990-2010 period were 7.3 kg CO_2-e / kg beef (grass-fed), 6.9 kg CO_2-e / kg beef (mid-fed grain finished) and 1.6 kg CO_2-e / kg beef (long-fed grain), or 6-34% higher than emissions excluding these sources. This analysis was strongly influenced by historic land use change in Australia which has declined rapidly in the past two decades. Emissions from LU and dLUC are expected to decline to between 0.4 to 0.8 kg CO_2-e / kg beef by 2026, showing a strong, annual reduction in these impacts.

Beef is a globally traded product, and concerns may exist regarding the impacts of transport on the environmental sustainability of beef. This study found that transport had a modest impact (<4%) on greenhouse gas emissions, suggesting that for red meat supply chains, transport distance is not a good indicator of greenhouse gas emissions or environmental impact. Importation to the East or West coast of the USA, or varying the transport distance to different inland cities had only a very small bearing on the results. Transport had a larger influence on energy demand (contributing about 20% of total energy) but whole supply chain levels remained comparatively low. Conversely, land occupation was generally higher because of the lower stocking rates used in the Australian rangelands compared to many Northern Hemisphere countries. Few data were available on water use for other regions in the world.

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1 Introduction

1.1 Background

Australia has been the second largest exporter of beef in the world for several years, closely following Brazil in total volume exported annually (FAO 2011). Australian beef is exported to many countries and regions, the largest of these being the USA, Japan, Korea and South Asia. Trade with the USA has traditionally focussed on lower grade meat for the processing industry. However, there is also an important and growing trade for Australian premium quality pasture fed and grain finished product into the USA. In the USA and around the world, retailers and consumers are seeking information regarding the provenance and sustainability of the products they consume. Concepts such as ‘food miles’ (Paxton 2011), or the transport distance involved in the food production system, have brought increased focus on the connection between transport distance and the sustainability of food. Studies of the US food industry have shown that transport distances involved in food production and supply can be considerable (Weber & Matthews 2008). However, the connection between transport distance and the sustainability of food production is less clear. Weber & Matthews (2008) conducted an input-output LCA on food consumption by U.S. households. The total freight (the transportation of one metric tonne a distance of 1 km, termed tonne kilometres or t.km) from production to retail for an average U.S. household was 12,000 t.km/household/yr, of which 3000 t.km (25%) was due to the final delivery of the food from the farm or production facility to the retail store (food miles). However, these authors found that transport throughout the supply chain contributed only 11% of the greenhouse gas (GHG) emissions associated with food production. Similarly, Ledgard et al. (2011) showed that for New Zealand lamb imported into Europe, the contribution of transport to total GHG was only 5% despite the very long transport distance (~17,000 km). These results suggest that a more holistic measure of the sustainability of food products is required. The preferred tool for conducting this analysis is life cycle assessment (LCA), as this tool investigates the whole production system to a specified end point (such as the point of retail) and can be used to investigate multiple impacts, such as GHG emissions, energy demand, consumptive water use and land occupation (Sala et al. 2013a, b).

To date, there has been no holistic analysis of Australian beef production, processing and transport to markets in the USA. The present study provides such an analysis, and aims to determine key aspects of the environmental sustainability of Australian beef production, processing and transport to the important USA export market for premium beef. While sustainability is a broad term that can encompass economic and social aspects, this study is focused on the key aspects of environmental sustainability; fossil fuel energy demand, consumptive and stress weighted water use, land occupation and GHG emissions.

1.2 Life Cycle Assessment

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

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

1.2.1 LCA research framework

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

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

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

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

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

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

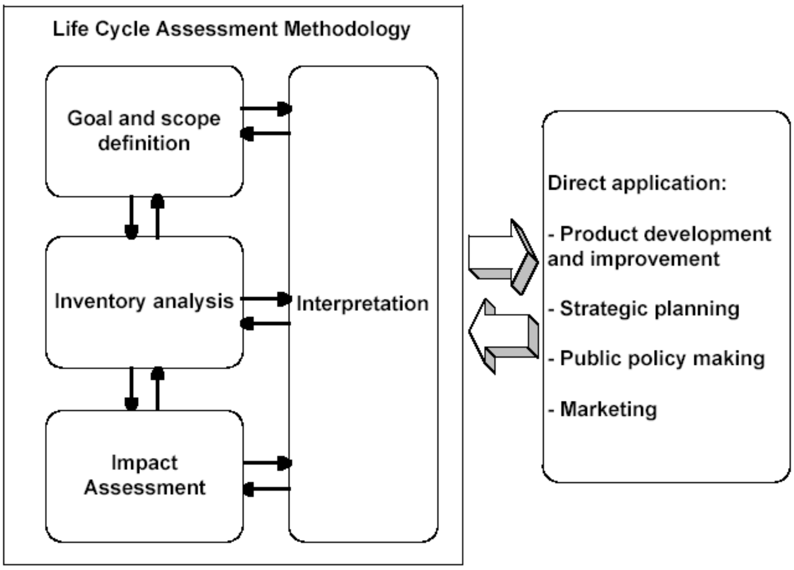

Figure 1 – General framework for LCA and its application (ISO 2006A: 14040)

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

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

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

1.2.2 Important methodological aspects of LCA research

Functional Units and System Boundaries

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

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

Inventory Development

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

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

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

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

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

Handling Co-Production

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

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

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

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

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

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

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

1.3 Sustainability in the beef industry

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

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

LCA has been increasingly seen as the state of the art relating to the environmental sustainability because of its holistic assessment through the supply chain, and also the multiple impact categories taken into account (Sala et al. 2013a, b). The focus of the present study is thus to use LCA to address some fundamentals of environmental sustainability in the beef industry, taking into account the key role that agriculture has in producing food for the world. The key elements of the investigation are therefore:

- Utilisation efficiency of key natural resources such as land, water and energy.
- Assessment of potential environmental impacts, specifically global warming and water stress.

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

Environmental impacts inevitably arise from production systems as a result of general operations. These impacts may damage any or all of the following; the resource base, the health of natural ecosystems or human health. In some instances the cause-effect relationship is clear. For example, phosphate losses from a farm can cause eutrophication (elevated nutrient levels) in a local river, leading to declining aquatic ecosystem health, changes in fish species or fish deaths. This may happen rapidly (i.e. in the space of days or weeks) and the result of improved practices may also be seen rapidly. On the other hand, the impacts of greenhouse gas emissions from a farm are less easily conceptualised. These impacts contribute to a global phenomenon with numerous causes and uncertain effects. Additionally, there is a very weak link between cause and effect at the local level, making it hard to ‘see’ the impact of emissions from a given farm. Nonetheless, such assessments must be made, because agriculture can have a significant contribution to overall impacts when whole industries (rather than individual farms) are taken into account.

It is possible to separately categorise resource utilisation (as a measure of the efficiency of food production) and environmental impact (negative or positive impacts arising from agricultural production). The former is more relevant in the discussion of food production and food security, while the latter is more relevant for the discussion of the on-going ability to produce food without adverse impacts on the resource base, other natural systems or human health.

The following sections outline the major resource efficiency issues (land, water and energy use) and environmental impact issues most relevant to beef production in Australia.

1.3.1 Fossil fuel energy demand

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

Energy use is less commonly assessed than GHG or water use but is nonetheless an important consideration with respect to resource use efficiency.

1.3.2 Consumptive and stress weighted water use

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

The Australian Bureau of Statistics reports one category that is specifically related to beef (irrigation water used for grazing meat cattle). Some other categories may contribute to water use in the supply chain (i.e. for the production of feed inputs for grazing or lot feeding). The ABS does not collect data relating to water use from farm dams, or water that livestock may directly consume from creeks or rivers. As a result, livestock drinking water supply is largely excluded from the ABS data. Australian water use data for a number of agricultural industries are presented in Figure 2.

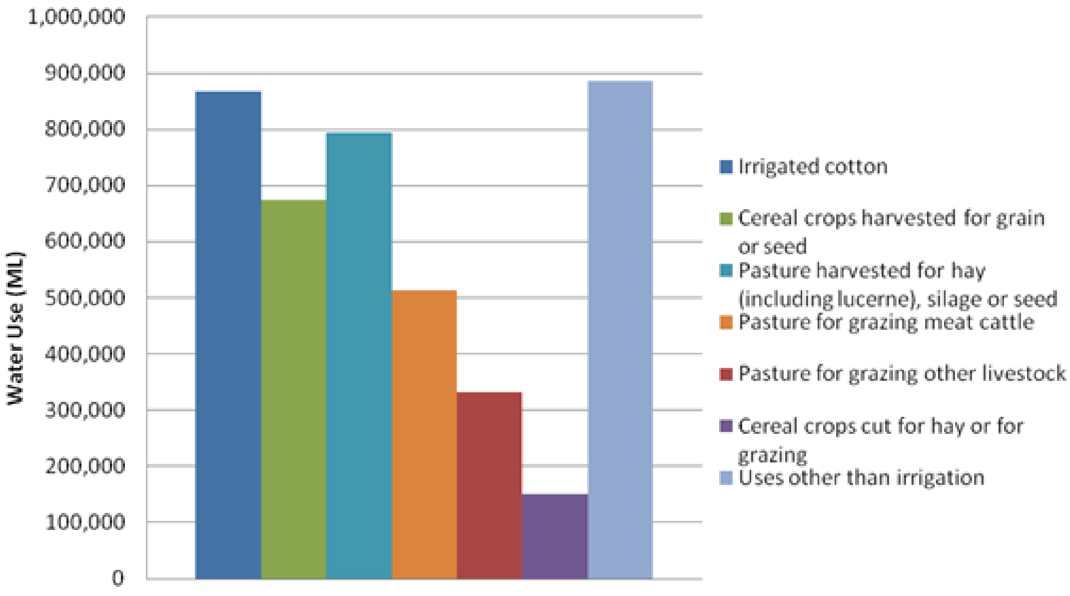

Figure 2 – Water requirements for a number of agricultural commodities (ABS 2008)

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

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

The term 'consumptive fresh water use' or simply 'consumptive water use' is a useful indicator of water use in volumetric terms. In an LCA context, this must include all consumptive 'uses' including losses, associated with the supply, which may be considerable. While this is broadly comparable to the term 'blue water' in the water footprinting literature (i.e. Hoekstra et al. 2011) it is not always clear in practice how comprehensive these studies are in estimating or including water supply losses. For example, methods for estimating these supply losses were not outlined by (Mekonnen & Hoekstra 2011, 2012) in their study of the water footprint of global livestock, though they may have been included.

The purpose of LCA is to investigate not simply the 'use' of a resource, but to determine the potential impact of that use. This is important for the discussion of water use. Consumptive water uses vary in their impact on other competitive users or the environment. Where water is plentiful, the relative stress on water reserves may be very low. Put simply, the '*the more you use, the worse you are'* principle is not a universally applicable concept for assessing water use. The impact of using water may be low, because there are sufficient volumes for all competitive users and sufficient volumes for maintaining aquatic ecosystem health at the current level of abstraction). To improve understanding of the impact of water use, impact assessment methods have been proposed by Mila i Canals et al.(2009) and Pfister et al.(2009). Pfister et al. (2009) described a method of determining the 'stress weighted' water use, by accounting for the expected impact of using water in a given catchment, using a global stress weighting factor. Ridoutt & Pfister (2010) further describe this method and apply the term 'stress-weighted water footprint', with units of L H_2O-e. The stress weighted water use impact assessment method applied different stress weighting factors for different regions of Australia. To calculate the stress weighted water use, consumptive water use in each region was multiplied by the relevant WSI and summed across the supply chain. The value was then divided by the global average WSI (0.602) and was expressed as water equivalents (H2O-e Ridoutt & Pfister 2010). Using this approach, Ridoutt et al. (2012a) estimated that the stress weighted water use for beef produced from a number of NSW production systems ranged from 3.3 – 221 L H_2O-e / kg LW.

1.3.3 Land occupation

Land resources are a limited global resource. Globally, of the total ice-free land surface of 13.4 billion hectares, approximately 3.5 billion ha (27%) are permanent pastures and 1.5 billion ha (12%) are under arable cultivation (Mercer 2013). With a growing demand for food and biofuel production from the world's land resources, utilisation efficiency is an increasingly important factor, though there is a general lack of consensus on how this should be measured in LCA. To date, most assessments have reported simply the total land required by a production system (i.e. for beef or pork or wheat) with no description of the type of land used, or the impact of using that land. Land types differ in productivity and suitability for cultivation and this needs to be taken into account in order to provide meaningful results.

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

Of the total land area of Australia (7.687 million km^2) only 7% is arable according to the (FAO 2008). However, at any given time closer to 3% is actually cultivated (ABARES 2010). Considering there are state regulations restricting conversion of pasture land to crop land, the total arable land may be closer to 3% than 7%. In contrast approximately 56% of Australia's land area is used for grazing livestock, mainly on native or naturalised pastures (Figure 3).

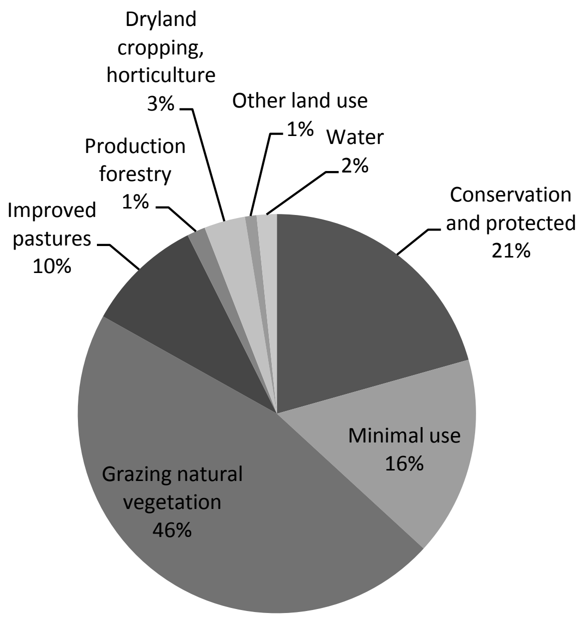

Figure 3 – Major land uses in Australia based on the 2005-06 dataset (ABARES 2010)

The vast majority of grazing land falls in the pastoral zone, which is generally unsuitable for other forms of agricultural production, particularly those reliant on cultivation, because of land and climate limitations. Land in the category "improved pastures" may be a combination of arable and non-arable land. However, because of regulatory constraints in some states (such as NSW), much of the pasture land that could be cultivated (from a land capability point of view) is restricted from conversion by legislation. In Australia, arable land used for cropping represents only 3.4% (0.26 M ha.) of total land mass (ABARES 2010). Consequently, this is a much more limiting resource and is subject to a much higher degree of competition for food production uses. The dominant competitive agricultural uses for arable land in Australia are grain (cereal and pulse) production, forage (crop) production for grazing animals and pasture production for grazing animals. It is informative therefore to investigate land occupation for different livestock systems in terms consistent with land capability and availability. While incomplete, it appears necessary to distinguish between arable and non-arable land types *at a minimum* when assessing land occupation from a resource perspective.

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

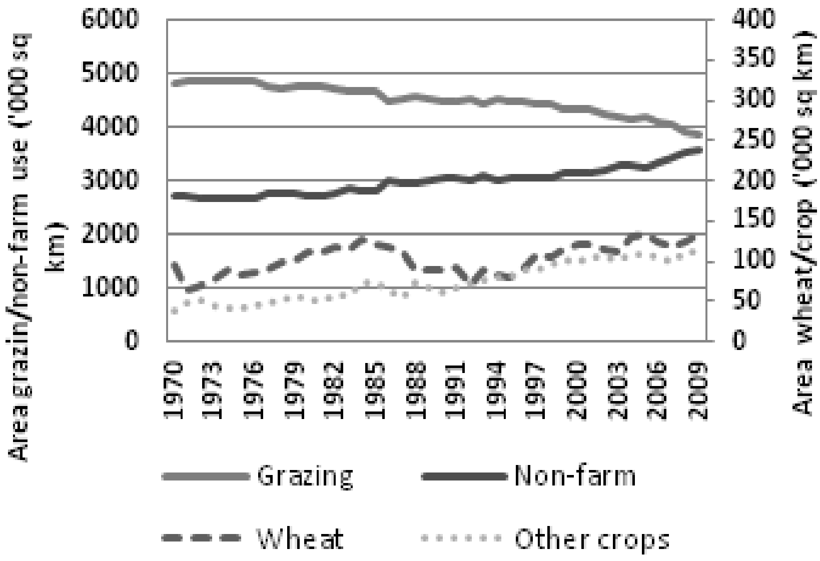

Figure 4 – Trends in land use for major agricultural production in Australia (ABARE 2009)

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

In the field of LCA, land occupation has most commonly been reported using a simple estimate of 'total land occupation' over a given time period, measured in square metres (m^2 yr). The extensive review of beef, pork, chicken, egg and milk LCA studies by de Vries & de Boer (2010) showed that beef production requires the greatest amount of land of all the livestock protein products, which is not surprising considering the differences in fecundity and feed conversion efficiency between the species. However, the authors were careful to note that this simple metric is not sufficient to make recommendations about which is the most 'efficient' meat product to produce. Ruminants (beef and sheep) can graze non-arable land, while pigs and poultry require arable land for feed production. They also note that poultry and pigs require grain which could be fed directly to humans, while red meat production may not. Clearly, total 'land use' is not very informative when discussing the efficiency of food production for ruminants; greater detail is required.

1.3.4 Land use change

There is potential to convert land from one land use to another, though this is constrained by biophysical factors including land type (soil, slope etc.), vegetation, annual rainfall, rainfall variability, evaporative loss, and by economic and policy restrictions. Land use mapping by the Australian Bureau of Rural Sciences (ABARES 2010) shows that in the five year period from 1996/97 to 2001/02, the area of land with natural vegetation used for agricultural production fell by 12.7 million ha nationally. This was largely due to an 11.6 million ha.

decline in grazing land. Most of the area of grazing land lost from production can be explained by increases in conservation reserves although there has also been a small increase in the area under cultivation. While all Australian states now have legislative restrictions on clearing of native vegetation, historic land clearing may still influence the GHG emissions attributable to livestock where this occurred less than 20 years ago (BSI 2011, ISO 2013). The present study included an analysis of impacts to global warming as a result of direct land use change (dLUC) where this occurred in the systems investigated.

1.3.5 Human edible protein efficiency (HEP-E)

Grain is an important primary commodity which can be used directly for human consumption or indirectly, via animal production systems. Australia is a major global grain producer and exporter. However, domestic consumption has increased rapidly over the past 10 years, primarily driven by increased consumption from livestock production (Spragg 2008). Livestock consumed an estimated 28% of grain produced in 2007 (Spragg 2008). The use of cereal grain for livestock feeding is important both from an environmental impact and a food security perspective, and is an important focus for research in both areas. The efficiency of utilisation of grain is an important consideration for the efficiency of livestock systems. Feed conversion ratio, or FCR, is a very important performance indicator for all livestock systems. There are marked differences between the species in terms of FCR; poultry are the most efficient, followed by pigs, then ruminants. Differences between the species arise from fundamental physiological differences. In particular, monogastrics (poultry and pigs) have a much more efficient digestive system for high starch (grain) diets. The monogastric species also have higher fecundity (more offspring per breeding animal) resulting in lower maintenance feed requirements for the breeding herd or flock. For example, breeding sows consume in the order of 55-65 kg feed / weaned pig, and produce 20-24 sale pigs per sow per year (see Wiedemann et al. 2012). In contrast, a beef cow may consume 3500 kg of feed per calf produced. It is also typical for beef herds to produce less than one calf per cow on average across a herd. At 75% weaning, the breeding herd may consume 4700 kg of feed per calf weaned, not accounting for the feed consumed by the calf. However, one very important difference exists. However, one very important difference exists. Ruminants consume grass, which has a very low level of digestibility for monogastric animals. Consequently, the whole herd/flock FCR is not comparable between monogastrics and ruminants, which consume mainly grass diets.

CAST (1999) reported the reported the ratio of human edible output from livestock products (energy and protein) to human edible input consumed by livestock as a way of quantifying the contribution or conflict between animal production and food supply. This metric, which could be termed the ‘human edible protein efficiency’, or HEP-E of a livestock system, is informative to the discussion of animal agriculture’s contribution to food supply. HEP-E is the inverse of human edible protein FCR. Thus, a HEP-E higher than 1.0 means that the production system consumes less human edible protein than it provides. Gill et al. (2010) noted this was an important factor in the discussion of livestock’s role in mitigating climate change in the context of food security. The human edible protein and energy efficiency for a number of species were reported by Gill et al. (2010) citing CAST (1999). These results are reproduced in part in Table 1, which shows the higher HEP-E of South Korean production, because of the higher use of forages rather than grain (for beef) compared to the USA.

Table 1 – Comparative efficiencies of different livestock production systems in terms of human edible energy and protein (Reproduced from Gill et al. 2010)

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

1.3.6 Greenhouse Gas Emissions

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

2 Methodology

2.1 Goal Definition

This project aimed to provide robust analysis of the sustainability of Australian beef exported to the USA. Specifically this report focuses on premium grass-fed and grain-fed beef for the retail and food sector markets in the USA. This product is predominantly sourced from the eastern states in Australia.

2.2 Project Scope

2.2.1 Functional Unit and System Boundary

The functional unit represents the primary output from the supply chain and is closely related to the system boundary. Results are presented per kilogram of boneless, retail ready Australian beef at the cold-storage warehouse in the USA. The system boundary includes all stages of production, and inputs, required to produce, process and transport Australian beef through to the point of cold storage on the east coast of the USA and included distribution to the point of retail (**Figure 5**). At this point, impacts through to consumption will not be greatly different regardless of the origin of the beef (i.e. from both cull and prime cattle).

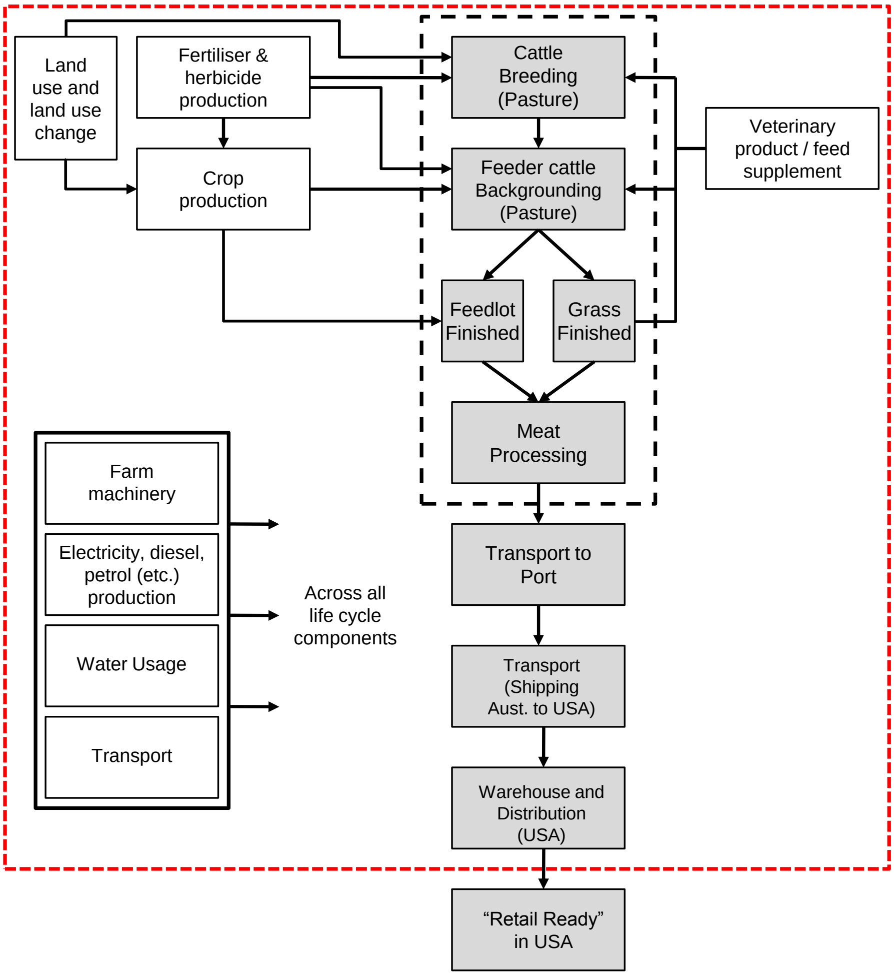

Figure 5 – Generalised system boundary for one kilogram of Australian beef produced and exported to the USA

2.3 Impact categories and methods

The study included assessment of five broad environmental impact and resource utilisation categories; energy demand, consumptive and stress weighted water use, land occupation, human edible protein efficiency, and greenhouse gas emissions. Methods are described in the following sections.

2.3.1 Fossil fuel energy demand

Primary energy was assessed using the fossil fuel energy demand (Goedkoop et al. 2009), measured in mega joules (MJ) using Lower Heating Values (LHV).

2.3.2 Consumptive and stress weighted water use

The water use inventory was developed using the Consumptive Fresh Water use (consumptive water use) indicator. Additionally, the impact assessment method 'stress weighted water use' was used (Pfister et al. 2009).

Table 2 – Water use classifications and methods

Water use reporting category	Units	Description	Noted exclusions
Consumptive Water Use (broadly analogous to blue water use)	L	All consumptive water uses throughout the supply chain including drinking water, water supply losses, evaporative losses from cleaning, and process water use. Return of urine was modelled as loss due to the large evaporation and the fact that it would not contribute to stream flow in an Australian context.	Degradative water uses were assessed to be relatively minor for the production systems of interest.
Stress weighted water use	L H_2O-e	All consumptive water uses multiplied by the relevant WSI value, summed across the supply chain and divided by the global average WSI (after Ridoutt et al. 2011).	

2.3.3 Land occupation

Land occupation has not previously been included in most Australian agricultural LCAs. Land occupation is a standard category within LCA and is a simple aggregation of the land area required to produce a given product. We have included agricultural land occupation, measured in m^2 yr) with three land occupation classifications; i) occupation of non-arable (rangelands) for pasture, ii) occupation of arable land – cultivated for grain or forage crop production, and iii) occupation of arable land for pasture.

At each farm, the proportion of land in each category was determined from information provided by the farmers and from field observations. Land areas were accurately determined using GIS software and aerial photography or satellite imagery. For each land occupation type, pasture production and utilisation rates were determined through discussion with the farmer and from stocking rate records. No characterisation factors were applied, and data were reported in m^2 of land occupied over a 12 month period.

2.3.4 Human edible protein efficiency (HEP-E)

The efficiency of human edible protein utilisation was modelled using a detailed inventory of grain use throughout the supply chain. Grains were characterised to determine the human edible protein (kg) content, taking into account milling losses where relevant. Human edible protein yield was determined from the boneless meat yield multiplied by a protein content of 0.19. Following the system expansion approach to handling co-products, we also modelled avoided human edible protein sources. For example, where beef co-products such as meat meal substitute for plant protein meals, this resulted in avoided human edible protein consumption.

2.3.5 Greenhouse gas emissions

Greenhouse gas emissions were determined from all sources relevant to beef production throughout the supply chain. The study applied IPCC AR4 global warming potentials (GWPs) of 25 for methane and 298 for nitrous oxide (Solomon et al. 2007). Emission prediction methods are outlined in Appendix 3 – Modelling GHG emissions.

2.3.6 Greenhouse gas emissions – Land use and direct land use change

The net GHG emissions associated with land use (LU) and direct land use change (dLUC) were estimated from regional averaged data. Net GHG emissions were not assessed on the case study farms (CSF) as land clearing had been negligible since 1990 on all but one farm and specific data were insufficient to present these results in a way that was consistent to the regional analysis.

A characteristic of Australian agriculture is that land development has occurred more recently than other developed countries, though clearing of remnant native vegetation for beef production has decreased markedly in the past two decades due to the introduction of government regulations. Another feature of the regions is that significant woody regrowth occurs in many areas previously cleared. An estimate of net GHG emissions due to change in forest and woodland cover was calculated from the loss of carbon stocks due to mechanical clearing and the carbon sequestration in growing woody vegetation. Areas of woody vegetation in the central-southern Queensland (QLD) and northern New South Wales (NSW) regions were assessed from satellite imagery analyses reported at the scale of Catchment Management Areas (CMAs) in NSW and bioregions for QLD. The Statewide Landcover and Trees Study (SLATS) uses world-leading methodology for remote sensing and ground-truthing of clearing rates for eastern Australia. Biomass of cleared remnant vegetation was estimated from the National Carbon Accounting Scheme (accessed through DCCEE (2013) with ratios for remnant to non-remnant clearing for each region estimated from SLATS data used to assign biomass for re-clearing of regrowth.

An estimation of the extent of woody regrowth was made based on SLATS reporting of total woody cover for each region and the area of remnant vegetation. This allowed a calculation to be made of carbon sequestration in regrowth and in turn the net emissions or removals from change in forest cover. Because of the high uncertainty in this estimate, two scenarios and two net GHG estimates were applied to explore the impact of LUC. Scenario 1 was based on net emissions from the period 1990-2010 and accounts for historic land clearing and regrowth in this time period. Scenario 2 investigates expected net emissions retrospectively from the year 2026. This represents the 20 year period after the introduction of land clearing regulations in the state of Queensland. The relative low rates of emissions for regions in NSW, and in Queensland since 2006, are of similar magnitude to, and often less than, the sequestration estimated to be occurring in previously cleared vegetation. Method details are presented in Appendix 4 – Land use change GHG methods and data.

Land use and dLUC emissions may also arise because of grain use for cattle production. Use of feed grain has increased over the past two decades and associated LU and dLUC emissions were assessed. Estimation of soil carbon loss or sequestration has a high uncertainty due to uncertainty in the impacts (both direction and magnitude of change) of management on soil carbon stocks. There is also uncertainty in the extent to which increased feed grain use is associated with increased yields rather than increased area of production and if the latter whether feed is grown on expanded area of production or on land previously cultivated for other crops. Three scenarios (low, medium, high loss of soil carbon) were run to produce a range reflecting the uncertainty in areas used to produce grain for beef feedlot usage and management practices. Method details are presented in Appendix 4 – Land use change GHG methods and data.

2.4 Inventory development

2.4.1 Collection of foreground data

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

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

Energy demand was determined from purchased energy (electricity, diesel, petrol) and transport records for purchased inputs used by the farm.

2.4.2 Modelling of foreground processes

Where data were not available for some inputs and outputs in the foreground system these were modelled or estimated from literature values. Key modelled inputs included drinking water use and feed intake (dry matter intake). These data were modelled from climate data, herd characteristics and livestock performance. Similarly, important outputs such as enteric methane emissions could not be measured, but were modelled based on the livestock herd.

2.4.3 Background data

Background data for upstream processes such as generation and supply of energy and purchased products such as fertiliser were sourced from the Australian LCI database (Life Cycle Strategies 2007). Energy demand associated with the manufacture of purchased inputs such as fertiliser was based on either the Australian LCI database (Life Cycle Strategies 2007) where available, or the European Ecoinvent (2.0) database (Frischknecht et al. 2005). Feed grain data were based on Wiedemann et al. (2010) and Wiedemann & McGahan (2011).

2.5 Supply chain characteristics

The majority of prime cattle produced for the USA premium grass-fed and grain-fed sectors are drawn from large production regions in the eastern states. This study was based on two major production regions; southern and central Queensland (QLD), and north and north western NSW. Collectively the central and southern regions in QLD represent ~25% of Australia's beef herd while the north and north-west regions of NSW represent a further 10% of the Australian herd.

Australian cattle exported for the premium markets in the USA are primarily steers finished on either grass or grain. The present study investigated beef bred in rangeland areas and exclusively pasture fed (grass fed), and steers finished on grain for either 115 days (Mid Fed – MF) or 330 days (Long Fed – LF). The Mid Fed category represents the most common class of grain finished export cattle in Australia. The Long Fed category is tailored to the production of a high quality beef product, predominantly from Angus or Wagyu breeds. A small amount of this product is exported to the USA for the restaurant trade. In the present study, we modelled cattle for the LF supply chain in NSW only, because this reflected production from the case study farms in this region more closely than in QLD.

For both regions, data were modelled from data collected from case study farms (CSF), and from a regional survey conducted by the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES), which were used to model a regional average farm (RAF) in this study.

2.5.1 Case study farms

Data were collected from a total of 15 case study farms and feedlots across the two regions. Farms were selected to represent typical production systems of two regions that supply the USA grass and grain-fed markets. The small dataset was necessary to conduct detailed modelling of water inputs and land occupation, but was a weakness with respect to comprehensive coverage of the production regions. This was addressed by augmenting the CSF dataset with regional survey data (see below) to improve representativeness. All farms were visited as part of the data collection process to ensure modelling of GHG and water were based on local conditions for each farm. Input data were collected to cover an average production cycle in the period 2007-2010. For critical herd productivity factors such as weaning percentage and growth rates in young cattle, results were averaged over a 3-5 year period of time to remove the effects of unusual seasons. Descriptions and abbreviations for the supply chains are provided in Table 3, and herd productivity data are provided in Table 4.

2.5.2 Regional average farms

To improve the representativeness of the case study farm dataset, ‘regional average’ farms were also developed using data from the Australian ABARES survey for regions 322, 321 and 331 in Queensland, and regions 121, 122 and 131 in NSW (ABARES 2013). Data from these surveys were extracted for the five year period to 2010. These data represent on average 115 beef producers across the two regions during the five year period, including both specialist and mixed enterprises. From these data, a herd model was constructed and inputs associated with beef production were determined. Growth rates for young cattle were not available from the survey. For both regions, we based growth rate assumptions on those provided in the Australian National Greenhouse Gas Inventory (NIR, 2012) for Queensland and NSW respectively. On review, the Queensland growth rates were reduced for young animals post weaning, to more closely reflect growth rates reported by Bortulussi et al. (2005). The survey did not provide data regarding drinking water supply sources, and these data were therefore based on data from the case study farms. Regional average data were not available for feedlots, so these data were based on the case study data collected as part of the project. Descriptions and abbreviations for the supply chains are provided in Table 3, and herd productivity data are provided in Table 4.

Table 3 – Description of supply chains modelled

Region	Cattle production description	Primary dataset	Abbreviation	Market
Central and Southern Queensland (QLD)	Pasture fed breeding, back grounding and finishing in extensive rangeland areas (grassfed)	Data collected from 5 Case Study Farms (CSF) Production modeled from a Regional Average Farm (RAF) based on survey data of 64 farms	QLD CSF Grassfed QLD RAF Grassfed	Premium retail or food sector
	Breeding and backgrounding on pasture in extensive rangeland areas, finished on grain for 115 days (MF).	Data collected from 9 Case Study Farms (CSF) and feedlots Production modeled from a Regional Average Farm (RAF) based on survey data for breeding farms, and case study data for feedlots	QLD CSF MF QLD RAF MF	Premium retail or food sector
Northern and North-western (NSW)	Pasture fed breeding, back grounding and finishing in more extensive and intensive rangeland areas (grassfed)	Data collected from 5 Case Study Farms (CSF) Production modeled from a Regional Average Farm (RAF) based on survey data of 51 farms	NSW CSF Grassfed NSW RAF Grassfed	Premium retail or food sector
	Breeding and backgrounding on pasture in extensive rangeland areas, finished on grain for 115 days (MF).	Data collected from 5 Case Study Farms and feedlots (CSF) Production modeled from a Regional Average Farm (RAF) based on survey data for breeding farms, and case study data for feedlots	NSW CSF MF NSW RAF MF	Premium retail or food sector
	Breeding and backgrounding on pasture in extensive rangeland areas, finished on grain for 330 days (LF).	Data collected from 5 Case Study Farms (CSF) Production modeled from a Regional Average Farm (RAF) based on survey data for breeding farms, and case study data for feedlots	NSW CSF LF NSW RAF LF	Premium market for niche, high quality meat from Wagyu or Wagyu cross cattle

Table 4 – Description of cattle production for the case study and regional average farms

Production parameter	Units	QLD CSF	QLD RAF	NSW CSF	NSW RAF
Weaning per cent	%	78.2	73.3	89.4	84.2
Breeder culling rate	%	19.9	20.0	14.6	15.0
Herd bulls	%	4.1	4.2	3.6	4.5
Mortality rate	%	2.4	1.8	1.5	2.3
Weaning weight	kg LW	228.7	216.2	237.0	204.7
Weaning age	months	7.4	7.0	8.0	8.0
Backgrounding					
Feedlot entry weight	kg LW	421.0	421.0	421.0	421.0
ADG (backgrounding)	kg / d	0.61	0.46	0.71	0.47
Grass Finished cattle					
Grass-finished steers	kg LW	577.8	548.6	573.8	557.2
Lifetime ADG (grass finished)	kg / d	0.56	0.55	0.64	0.52
Grain Finished cattle					
Grain finished steers - MF	kg LW	622.0	622.0	622.0	622.0
Grain finished steers - LF	kg LW	n.a.	n.a.	761.0	761.0
Lifetime ADG (grain finished - MF)	kg / d	0.79	0.76	0.94	0.72
Lifetime ADG (grain finished - LF)	kg / d	n.a.	n.a.	0.86	0.70

2.5.3 Meat processing, transport and storage

Primary data were collected from two major meat processing plants in QLD, and data were supplemented by a recent survey of resource use from beef processing plants (GHD 2011). Effluent treatment emissions were modelled from primary data collected from the processing plants.

Transport stages were included throughout the supply chain based on representative truck types and load specifications. International transport of chilled boneless beef was via ocean-liner to the USA. Product was assumed to be imported to the port of Philadelphia as a conservative estimate of total transport distance. The impact of importing to a closer port (Los Angeles) was also investigated. Transport and warehousing in the USA were modelled from a review of beef import processes and interviews with importers. Detailed inventory data are supplied in Appendix 1.

2.6 Handling co-production

There were a number of points in the production system where co-products are produced, and a method is required to divide burdens between products. In some cases, cattle farms may produce other products along with beef (sheep, grain) and the impacts associated with these must be separated. Methods are described in the following sections.

2.6.1 Dividing production systems

We handled co-production of beef, sheep and cereal grain on the farms by subdividing the farm into systems and accounting for each separately. This was achieved by dividing specific inputs and animal/plant processes between the systems. For inputs that were not specific to a particular sub-system, such as administration overheads or fertiliser inputs to pasture consumed by both sheep and cattle, these were divided based on the utilisation of land resources.

2.6.2 Co-production in the beef system

Within the beef production system, there are a range of products that are generated at different points in the supply chain. The supply chains produced beef from culled breeding cows, surplus heifers, and steers. In general, the premium beef supplied to the USA market is from steers. Only particular cuts are suitable for this market, such as tenderloin, striploin, rib eye and rump. Consequently, a considerable amount of meat, including edible offal, is diverted to other markets. In this project, impacts were divided evenly over all human edible products from the beef supply chain because there are no significant biophysical or nutritional differences between the products.

At the point of meat processing, impacts were also divided between meat and hides based on protein content in the products. Other products from meat processing such as meat meal and tallow were handled using system expansion. Similarly, manure production from the feedlot was handled using system expansion. Decisions regarding co-production are described in Table 5.

Table 5 – Methods for handling co-production

Stage in Supply Chain	Product and co-product (in brackets)	Method	Reason for choosing method for handling co-production
Feed inputs	Multiple	Economic allocation	Economic allocation is easily applied for minor background processes.
Grazing farm	Sale cattle (cull cows)	No allocation applied	There was no clear rationale for discriminating between beef from prime and cull cattle, considering the meat product from both classes of cattle is suitable for human consumption. Functional differences relate to markets and consumer preferences but not nutritional quality. The output from all systems was taken to be total beef produced from all classes of saleable cattle.
Feedlot	Beef live weight (nutrients contained in manure).	System expansion	Where manure nutrients were used to replace synthetic fertilisers, a system expansion process was used.
Meat Processing	Boneless meat products (hides, meat and bone meal, tallow)	Biophysical allocation and system expansion	Allocation between meat products and hides performed using a biophysical allocation method based on protein content. System expansion was used to account for by-products which are primarily used as animal feeds or pet food.

Meat processing yields and allocation factors are shown in Table 6. These values were collected from meat processing plants in Australia and correspond to a carcase yield of 53% and a retail yield (carcase to boneless beef) of 74%.

Table 6 – Meat products and co-products per 1000 kilograms of live weight beef processed

Products	Mass of product (kg)	Economic Allocation Factors	System expansion substitution products
Boneless meat products	433	89.9%	
Hides	65	4.8%	
Meat, blood and bone meal	84	2.2%	Soymeal and sorghum– on protein and energy equiv. basis
Tallow	86	2.8%	canola oil
Pet food	6	0.2%	Soymeal and sorghum– on protein and energy equiv. basis
Totals	674	100%	

3 Results

Beef exported to the USA is not differentiated by region of origin. Market requirements specify high weight-for-age which favours beef from southern regions where climate is more reliable, though the exact ratios are not known. We have assumed that grass-fed and mid-fed grain-finished cattle are supplied 65% from the NSW regions and 35% from the QLD regions assessed, while long-fed cattle are supplied from NSW only.

3.1 Resource use

3.1.1 Energy demand

Total fossil fuel demand from the QLD and NSW supply chains ranged from 24.2 to 44.3 MJ per kg boneless beef (Figure 6). The largest contribution was from the farm (averaging 59%), followed by meat processing (averaging 23%) and transportation of meat from Australia to US warehouse (averaging 20%). Grain-finished supply chains were found to have higher energy intensity than grassland-finished supply chains.

There was a trend towards higher energy use for the RAF systems, which corresponded to slightly higher levels of farm inputs compared to the case study farms.

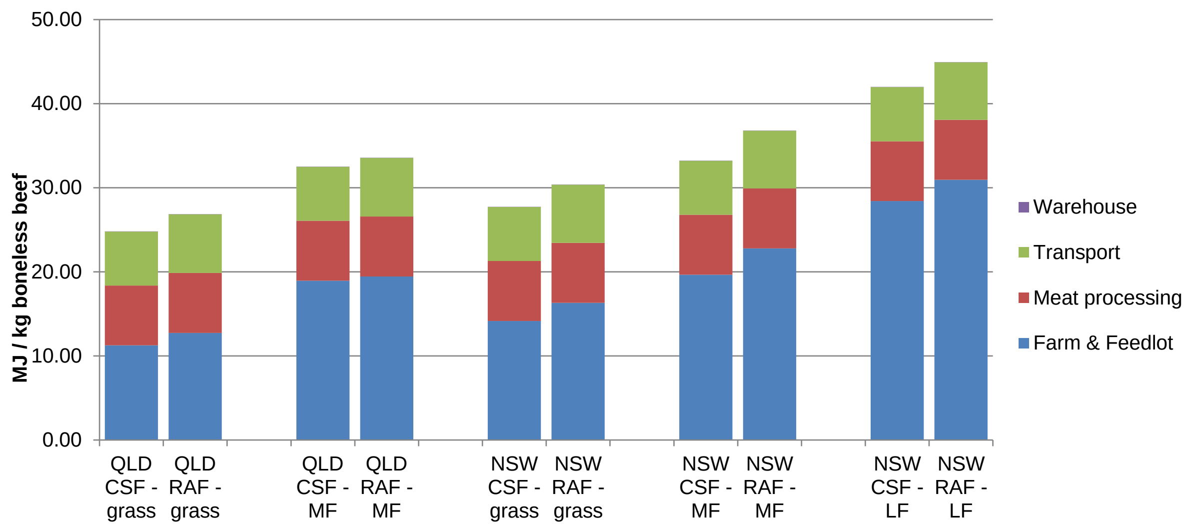

Figure 6 – Contribution of processes to fossil energy per kg of boneless beef from central and southern Queensland, north and north-west NSW long-fed (LF) grain, medium-fed (MF) grain and grass-fed supply chains. CSF = case study farms, RAF = regional average farms

3.1.2 Consumptive fresh water use

Consumptive water use from the QLD and NSW supply chains ranged from 410.2 to 640.3 L per kg boneless beef (Figure 7). Consumptive water use was negligible from the transport and warehousing stages of the supply chain and thus the contributions were only presented up to processing stage. Drinking water supply loss (at farm and feedlot) was the largest contributor (averaging 49%), followed by livestock drinking water (averaging 27%), and irrigation (averaging 18%).

Variation in irrigation and associated water use was found between case study farms and regional average farms of grass-finished supply chains. Case study farm data, while providing a reasonable representation of herd productivity and inputs, were less representative with regards to irrigation. None of the case study farms in Queensland had irrigation supplies; consequently water use was higher from the regional average dataset which included the small proportion of irrigation water use across the region. The NSW grass-fed results were reversed; irrigation water use was higher from the case study dataset, where one out of five farms used irrigation, than in the regional average dataset which was based on a larger number of randomly selected farms with lower overall irrigation.

Irrigation water use in the grain fed supply chains was mostly related to feedlot ration inputs. Both the case study and regional average datasets used the same rations, which were based on grain market data for the north eastern Australian grain growing region.

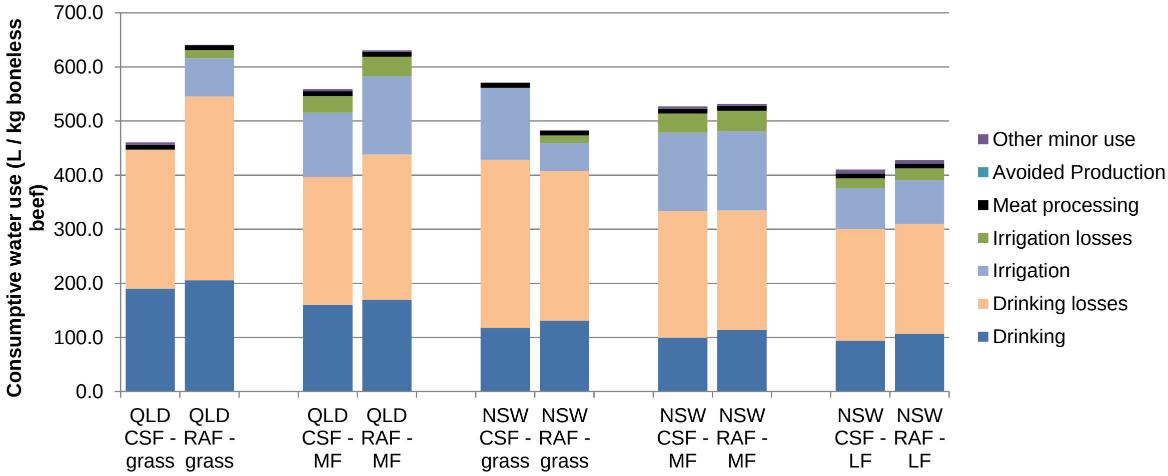

Figure 7 – Contribution of processes to consumptive water use per kg of boneless beef from central and southern Queensland, north and north-west NSW long-fed (LF) grain, medium-fed (MF) grain and grass-fed supply chains. CSF = case study farms, RAF = regional average farms

3.1.3 Land occupation

Cultivated arable land occupation through the supply chain varied from 0.0 to 28.7 m^2 per kg boneless beef, while total land occupation varied from 94.1 to 684.2 m^2 per kg boneless beef (Table 7). Grain production was the predominant contributor to cultivated arable land occupation. Consequently, grain-finished systems occupied more cultivated arable land than grass-finished systems. However, grain-finished systems were more efficient, therefore utilising less total land area than the grass finished systems. The much higher land use in the QLD supply chains compared to NSW corresponds to much lower stocking densities in these regions, which corresponds to lower rainfall and higher evapo-transpiration rates.

Table 7 – Land occupation per kg of boneless beef from central and southern Queensland, north and north-west NSW long-fed (LF) grain, medium-fed (MF) grain and grass-fed supply chains. CSF = case study farms, RAF = regional average farms

	Total land occupation	Cultivated arable land	Arable pasture land*
			m^2/kg boneless beef
QLD CSF - grass	653.1	0.0	6.2
QLD RAF - grass	684.2	2.8	34.1
QLD CSF - MF	532.7	12.2	4.9
QLD RAF - MF	521.0	14.0	25.4
NSW CSF - grass	111.1	9.6	20.5
NSW RAF - grass	242.9	2.4	12.0
NSW CSF - MF	94.1	16.3	16.0
NSW RAF - MF	192.2	15.6	8.8
NSW CSF - LF	97.3	28.7	15.3
NSW RAF - LF	183.6	27.7	7.8

* Land used for pasture but suitable for production of arable crops

3.1.4 Human edible protein efficiency

Human edible protein efficiency (HEP-E), i.e. the ratio of HEP in beef relative to HEP consumed) ranged from 1.8-12.3 for the NSW grass finished to 11.0-107.8 for the Queensland grass finished beef. The high ratio values corresponded to very low levels of human edible protein use on the Queensland grass-fed case study farms. Some of these systems supplement cattle with a non-protein-nitrogen source such as urea in some seasons but rarely supplement with grain. The NSW regions were more intensive and utilised small amounts of grain, leading to much lower ratios.

The grain fed systems had lower human edible protein efficiencies, ranging from 0.6-0.9 for the mid-fed scenarios, and 0.4-0.5 for the long fed scenarios. There was a strong trend towards higher efficiencies in the Queensland regions compared to NSW, and higher efficiencies for grass compared to grain fed.

3.2 Environmental impacts

3.2.1 Stress weighted water use

Consumptive water use is informative for assessing resource use, but less informative for assessing environmental impacts because the impact of using water varies depending on the level of stress of the water resource. This study applied the global water stress index of Pfister et al. (2009) to provide an indication of the impact of using water resources in the different regions and production systems. Across all supply chains, stress weighted water use was considerably lower than consumptive water use, showing that the impact of using water in these regions is considerably lower than the global average.

Stress weighted water use results differed considerably between regions and supply chains. Approximately 20% of cattle in the Queensland regions were produced in zones of high water stress with the remainder produced in zones of low water stress. In contrast, the NSW regions had lower levels of water stress, which led to lower stress weighted water use across all NSW supply chains (Figure 8).

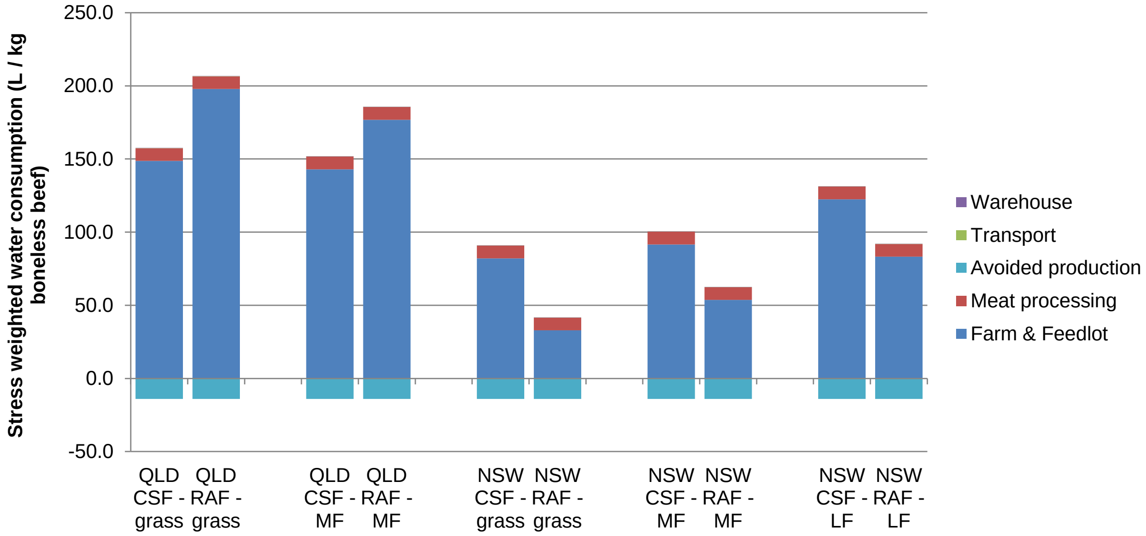

Figure 8 – Contribution of processes to stress weighted water use per kg of boneless beef from central and southern Queensland, north and north-west NSW long-fed (LF) grain, medium-fed (MF) grain and grass-fed supply chains. CSF = case study farms, RAF = regional average farms

3.2.2 Greenhouse gas emissions

Greenhouse gas emissions from the QLD and NSW supply chains ranged from 19.8 to 27.1 kg CO_2 e per kg boneless beef (Figure 9). Contributions of components were similar between the supply chains. The predominant contribution was from primary production (av. 93%), followed by meat processing (4%) and transportation of meat from Australia to the warehouse in the USA (3%). By source, enteric methane was the single largest emission (av. 54% to 84%) followed by carbon dioxide emissions from energy (9% to 13%), and manure nitrous oxide (6% to 13%). Herd productivity factors (weaning percentage, ADG to finished weight) were the major drivers of enteric methane emissions and therefore, GHG intensity. In both regions, there was a trend towards higher emissions for the regional average farms compared to the case study farms, which related to lower weaning rates and steer weights at turnoff for the RAF analysis.

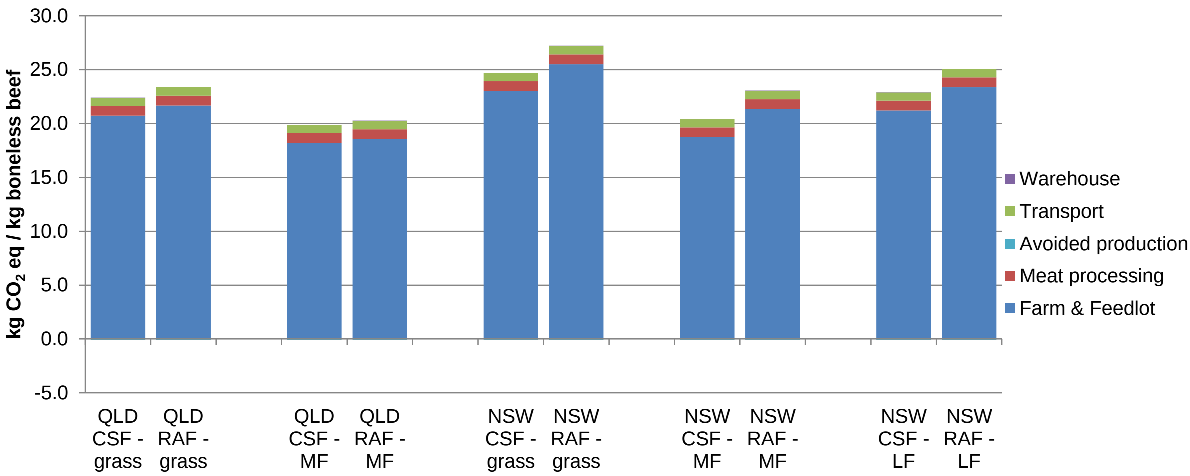

Figure 9 – Contribution of processes to GHG emissions per kg of boneless beef from central and southern Queensland, north and north-west NSW long-fed (LF) grain, medium-fed (MF) grain and grass-fed supply chains. CSF = case study farms, RAF = regional average farms

It should be noted that the major difference in estimated farm level GWP between the QLD and NSW regions was driven by the use of different enteric methane prediction methods, which resulted in higher levels of predicted enteric methane from the NSW supply chains than the Queensland supply chains. The sensitivity of results to the enteric methane prediction method is examined in the next section.

3.2.3 GHG emissions from land use and direct land use change

Emissions from LU and dLUC were subject to a greater degree of uncertainty in the emissions estimation. To account for this, a low and high emission scenario approach was applied and results were averaged to determine mean emissions. Emissions for beef produced in the period 1990-2010 and for the period 2006-2026 were determined, corresponding to two periods that differ substantially in regulations controlling deforestation in Australia.

Averaged across the QLD and NSW regions, emissions over the 1990-2010 period were 7.3 (4.1-10.6) kg CO_2-e / kg beef (grass-fed), 6.9 (4.5-9.3) kg CO_2-e / kg beef (mid-fed grain finished) and 1.6 (0.4-2.7) kg CO_2-e / kg beef (long-fed grain). Deforestation for pasture was higher in Queensland than NSW. One product (long-fed grain finished beef) was drawn from the NSW region only resulting in substantially lower emissions than beef products drawn from both regions.

4 Discussion

This study represents the first comprehensive LCA of Australian beef supply chains from paddock to the warehouse in the USA, covering multiple impacts relevant to the industry.

To aid interpretation of the results, a sensitivity analysis was performed to investigate the impact of alternative assumptions throughout the model.

4.1 Sensitivity analysis

4.1.1 Handling co-production

Impact and resource use results are sensitive to the methods applied for handling co-production. In the present study, a range of methods were applied to handle co-production throughout the supply chain. In order to examine the effect of the most sensitive choices, we applied an alternate approach; allocation on the basis of the economic value of products for the meat processing stage. Results from the two methods differed depending on category. Greenhouse gas and energy results were 7 to 9% higher using economic allocation at the meat processing stage. Consumptive water use was 6 to 9% higher using the economic allocation, because there was a substantial reduction in water use associated with avoided irrigation in the soymeal processes substituted for meat meal.

Numerous livestock studies have also presented results on a carcase weight basis ‘at the farm-gate’ by simply applying a dressing percentage value to convert live-weight results to carcase weight. This approach is inaccurate, as it fails to take into account the added impacts associated with transport and processing of cattle to produce meat. It also fails to take into account that cattle produce valuable co-products such as hides. In the present study, including all impacts and co-products associated with meat processing resulted in ±34% difference across impacts for the meat product, compared to using farm gate results and attributing all impacts to meat, using standard carcase and retail yield values.

4.1.2 GHG model assumptions

Within the GHG prediction model, enteric methane was the largest emission source for beef production in all supply chains. This study applied regionally representative methods to predict enteric methane for the Queensland beef supply chains (Kennedy & Charmley 2012) and a more generalised model for predicting enteric methane from the NSW supply chains (Blaxter & Clapperton 1965) and lot-feeding (Moe & Tyrrell 1979) which are applied in the Australian NGGI. In order to test these assumptions, we compared these with the IPCC 2006 method (Dong et al. 2006), which assumes methane yield is 6.5% ± 1.0% of gross energy intake (GEI) for cattle grazing pastures, and 3.0% ± 1.0% for lot-fed cattle. Application of the Kennedy & Charmley (2012) model resulted in slightly lower enteric methane predictions (5.9% GEI) compared to the default IPCC 2006 method. Conversely, application of the Blaxter & Clapperton (1965) model resulted in higher predicted enteric methane (av. 7.2-7.6% GEI) than the IPCC 2006 default. The consequence of this was that emissions from the QLD grass-fed supply chain were 9% lower than would be predicted using the IPCC default. Conversely, the NSW results were 9% higher than if the IPCC default was applied. The effects were less significant for the mid-fed feedlot supply chains, where predicted enteric methane emissions from feedlot cattle where similar to the IPCC 2006 recommended value of 3.0% ± 1.0%. The long-fed feedlot cattle, which consume a diet with higher proportions of roughage, had predicted enteric methane levels of 6.8% GEI when using Moe & Tyrell (1979).

This sensitivity analysis does not imply inaccuracy in the prediction models applied in the study; the prediction equation from Kennedy & Charmley (2012) was developed for similar genotype cattle, grazing similar pastures to those in the Queensland regions assessed. Considering these findings, differences between QLD and NSW could not be confidently stated, as these may relate to the methodological differences rather than actual differences in emissions.

4.1.3 Transportation to the USA

Australian beef is imported into several ports, with the largest volumes arriving at the ports of Philadelphia and Los Angeles. Ocean transport distances are longer to Philadelphia (~18,000 km) compared to ~12,000 km to Los Angeles. As a conservative estimate, we presented results for importation via the port of Philadelphia. Transport distances in the USA were based on 50% of supply being transported 200 km and 50% being transported 1000 km. Importing beef into the port of LA had a negligible effect on GHG (-0.3%) and a modest impact on energy use (-4%). These results were not unexpected considering the small contribution from transport found by this study.

4.1.4 Land use and direct land use change GHG emissions

Emissions from LU and dLUC were very sensitive to the method applied both in determining emissions and attributing these to beef production. Uncertainty in the prediction methods is shown in the range of values provided. Emissions were attributed to beef accounting for all emissions since 1990 annualised over each year in that time period. Australia introduced legislation to control land clearing progressively from 1996 (NSW) through to 2006 (QLD) which have had the effect of dramatically reducing emissions over the last two decades. Because of the historic method applied here, annual emissions from dLUC decline annually and are expected to reach negligible levels by 2026. To demonstrate this, we modelled emissions retrospectively from 2026. This analysis showed emissions declining to 0.4 (-2.8 to 3.7) kg CO_2-e per kilogram of grass-fed beef and 0.8 (-1.6 to 3.2) kg CO_2-e per kilogram of mid-fed grain finished beef. Emissions from long-fed beef were 0.7 (-0.4 to 1.8) kg CO_2-e / kg beef. This analysis shows the complexity of understanding dLUC results that are based on retrospective methods where there has been a clear change in policy regarding land clearing, as is the case in Australia. Current historic emissions reported here are heavily influenced by land clearing 2 decades ago, while emissions in the past 8 years are substantially and demonstrably lower (DCCEE 2012b).

4.2 Comparison with the literature

4.2.1 Australian studies

Comparison of results from LCA studies is complicated by differences in the methods applied, the scope and boundaries of the systems. A number of published and unpublished LCA studies have been completed for Australian beef cattle. Most of these studies investigated the supply chain to the farm-gate only and presented results on a 'per kilogram of live weight' basis. We explored the consensus between previous studies and the present at the 'farm gate' level, per kilogram of live-weight produced immediately prior to transport for processing. To do this, results 'per kilogram of live weight at the farm gate' are presented in Table 8.

Table 8 – Impacts per kilogram of live weight produced at the farm gate for grass and grain fed beef produced from central and southern QLD and north and north-western NSW

		QLD CSF Grass	QLD RAF Grass	QLD CSF MF	QLD RAF MF	NSW CSF Grass	NSW RAF Grass	NSW CSF MF	NSW RAF MF	NSW CSF LF	NSW RAF LF
Global Warming	kg CO_2-e	10.9	11.4	9.6	9.8	12.1	12.2	9.9	11.2	11.2	12.3
Fossil energy	MJ	5.9	6.8	10.0	10.3	7.4	8.7	10.3	12.2	14.9	16.5
Consumptive Water Use	L	237.3	322.9	280.4	318.9	295.4	248.9	272.1	274.7	210.8	220.3
Stress weighted water use	L H_2O-e	78.2	101.3	74.9	90.5	43.2	17.3	48.2	28.2	64.4	43.8
Cultivated arable land	m^2	0.2	1.7	6.6	7.6	5.2	1.4	8.8	8.4	15.3	14.8
Total Land Occupation	m^2	343.7	360.0	280.3	274.2	58.6	128.0	49.7	101.3	51.4	96.8

For Queensland grass-fed steer production, results from the present study per kilogram of live weight were lower than previous studies for Queensland grass-fed steers reported by Wiedemann et al. (2013) of 12.9 kg CO_2-e / kg LW, or those reported by Eady et al. (2011) of 14.5 kg CO_2-e / kg LW. Eady et al. (2011) applied different GWP values and a different enteric methane prediction model in their study. When reanalysed using the same methods as the present study the results were 12.3 kg CO_2-e / kg LW. The lower impacts for case study farms in the present study relate to better productivity (weaning rates and growth rates) than the production system reported by Wiedemann et al. (2013b). Herd productivity was similar, though growth rates may have been lower, compared to the system studied by Eady et al. (2011). Eady et al. (2011) applied an economic allocation process which resulted in slightly higher impacts being attributed to the steers than the present study. The present study focussed on production for the USA premium grass and grain-fed markets, which preferentially select for younger, heavier cattle. Consequently, the results presented are representative of cattle exported to this market, but may have lower impacts than the average beef production in Queensland.

For NSW beef cattle, Wiedemann et al.(2013a) reported GHG emissions of 10.3-13.0 kg CO_2-e / kg LW for similar classes of steers. Ridoutt et al. (2012a) reported similar to lower emissions of 10.2-10.8 kg CO_2-e/kg LW for theoretical production supply chains with similar classes of steers in NSW. Peters et al. (2010a), Peters et al. (2010b) reported lower values of 9.2-11 kg CO_2-e/kg LW (adjusted to standardise the GWP values). Each of these studies used the same enteric methane model. Differences in impacts between the studies relate to differences production efficiency of the supply chain or to differences in the system boundary. No other study to the author's knowledge has comprehensively accounted for LU and dLUC emissions from Australian beef.

Consumptive water use results were higher than previously reported by Ridoutt et al. (2012b) who reported water use of 25–234 L / kg LW for beef from six theoretical production systems in NSW, compared to 210.8-322.9 L / kg LW in the present study. When compared using meat processing assumptions from the present study, results from Ridoutt et al. (2012b) were 49-465 L / kg boneless beef, compared to 410.2 to 640.3 L / kg boneless beef presented here. The primary reason for the higher water use in the present study was the higher estimated supply losses for livestock drinking water and irrigation water. In the present study, these estimated losses were the single largest source of water use, contributing 49% or 255.3 L on average across the supply chains. This was much higher than estimated by Ridoutt et al. (2012b) partly because of the different estimation of dam water demand and evaporation losses in the two studies. Water use associated with irrigated pasture and purchased feed inputs were also higher across the supply chains reported here, but were similar to the higher estimate reported by Ridoutt et al. (2012b) for their theoretical beef production systems.

4.2.2 Comparison with international studies

To improve the comparability of the results presented here with the literature, we have analysed results from a number of European and North American studies using standardised GWP values and standardised meat processing assumptions, including both the impacts (inputs, emissions) from meat processing, and taking co-products into account.

Greenhouse gas and energy demand results from the present study ranged from 19.8-27.1 kg CO_2-e and 24.2 to 44.3 MJ / kg boneless beef, including transport and warehousing impacts. Greenhouse gas emissions from European studies that investigated 'purpose grown' (i.e. non-dairy) beef production were in the order of 26.9-33.3 kg CO_2-e / kg boneless beef at the processor gate when assumptions from the present study were applied. Casey & Holden (2006) reported GHG intensity of 12.2-14.3 kg CO_2-e / kg LW for Irish beef production when GWP values were standardised. This corresponded to ~26.9-31.6 kg CO_2-e / kg boneless beef. Williams et al. (2006) reported GHG impacts of UK purpose grown beef of 25.3 kg CO_2-e / kg carcass weight, which converted to ~32 kg CO_2-e / kg boneless beef with meat processing and standardised GWP values. Energy use from Williams et al. (2006) was in the order of 25.3 MJ / kg carcass mass, which corresponded to ~54 MJ / kg boneless beef. Nguyen et al. (2012) reported GHG emissions of 27.0-27.9 kg CO_2-e / kg of carcass mass, or 33.3-34.5 kg CO_2-e / kg of boneless beef, and energy use of 64.8-73.4 MJ / kg carcass mass, or ~83-95 MJ / kg of boneless beef, for beef production systems from France.

For studies of North American beef, GHG emissions ranged from 27.8-41.5 kg CO_2-e / kg boneless beef using the same meat processing assumptions as the present study. Emissions from lot-fed beef in Canada were 13.0 kg CO_2-e / kg LW, or 30.3 kg CO_2-e / kg boneless beef (Beauchemin et al. 2010). Pelletier et al. (2010) reported results from beef production in the mid-west of the USA. This study showed GHG emissions of 14.8-16.2 kg CO_2-e / kg LW for feedlot beef with different backgrounding and grain feeding periods, which was 32.2-35.2 kg CO_2-e / kg when converted to boneless beef. Emissions from grass-finished cattle were reported as 19.2 kg CO_2-e / kg LW (41.5 kg CO_2-e / kg boneless beef) (Pelletier et al. 2010). Pelletier et al. (2010) suggested emissions may be substantially lower if soil carbon sequestration from grazed pastures was taken into account. Lupo et al. (2013) reported GHG impacts of 23.0 kg CO_2-e per kg carcass weight, or 27.8 kg CO_2-e / kg boneless beef, for feedlot finished beef produced on the northern great plains in the USA. Energy use ranged from 87-109 MJ / kg boneless beef (adapted from Pelletier et al. 2010). Fewer studies have been conducted for South American beef production. Cederberg et al.

(2009) reported a national average emission for Brazilian beef of 23-34 kg CO_2e kg CW (LUC not included) and corresponding values of ~34 kg CO_2-e / kg of boneless beef when consistent allocation methods were applied.

GHG emissions from the Australian beef supply chains were comparable to previous Australian studies, but tended to be lower than studies of European or North American beef production when LU and dLUC were excluded. The largest difference between impacts from Australian beef and beef from the northern hemisphere was the lower nitrous oxide emissions from manure and feed production in Australia. Australian conditions do not favour nitrous oxide emissions; rainfall tends to be lower (500-750 mm in the main beef production regions), and evaporation is very high, exceeding 2000 mm in some regions. Consequently, soil conditions are dry and nitrous oxide emissions are lower than may be expected in wetter climates. This is reflected in the lower nitrous oxide emission factors for manure and fertiliser in Australia than for European or North American conditions. The extensive production systems in Australia use low inputs of fertiliser and fossil fuel. This partly contributes to low GHG levels, and led to much lower energy use than most studies in the Northern Hemisphere. Fewer international studies have accounted for dLUC, though this is now understood to be an important factor in LCA/carbon footprint research (BSI 2011, ISO 2013). With LU and dLUC emissions included, average grass fed and grain finished beef from Australia were comparable to several European and North American studies that do not include dLUC emissions. Considering emissions from this source are declining substantially in Australia because of legislated controls on deforestation, future emissions are expected to decline to very low levels by 2026.

Land occupation could only be compared as 'totals' which are of limited value. As expected, land occupation was much higher in the present study (94.1 to 684.2 m^2 / kg boneless beef) than most studies in the literature, in response to the low stocking densities typical of Australian production. Land occupation was lower in Europe (~47-55 m^2 / kg boneless beef, Nguyen et al. 2012, Williams et al. 2006). No studies were found that reported land occupation in the USA. Assessment of total land occupation is not informative of environmental impacts or resource use efficiency in Australian conditions for a number of reasons. Firstly, total land occupation offers little insight into the resource value of this land, particularly when compared to other potential uses. We preferred to differentiate between arable and non-arable land resources in order to provide results that could be more meaningfully compared with other protein products that rely on arable land. It should also be noted that high levels of total land occupation do not imply greater environmental impacts necessarily. In many cases, higher land occupation (low stocking densities) will result in better ecosystem outcomes in rangeland regions than lower land use and higher stocking rates. Consequently, higher land occupation in the rangeland areas may be seen as preferable.

For consumptive water use, most studies available in the literature applied different system boundaries and may not have included losses associated with water supply for livestock drinking and irrigation. While not full life cycle assessments of consumptive water use, results from Becket and Oltjen (1993) were 3682 L / kg boneless beef for beef production in the USA. Capper (2011) reported water use of 1763 L / kilogram of carcass weight at the farm gate, which would be closer to 2300 L / kg boneless beef. The higher water use from these studies compared to those presented here (524.1 L / kg boneless beef) relates to the lower irrigation water use in Australia compared to the analysis made by these authors for the USA.

Zonderland-Thomassen et al. (2013) assessed water scarcity of beef production in New Zealand farms and found an average of 0.37 L H_2O-eq / kg LW, which is much lower than the stress weighted water use in the present study (17.3-101.3 L H_2O-eq / kg LW). This is understandable because of the higher water stress levels in this study (0.02-0.85) than that used by Zonderland-Thomassen et al. (2013) (on average 0.01). The warmer climate in Australia may also contributed to the higher water consumption and evaporation. However, it was not clear how the water consumption was modelled and thus detailed comparison was not possible.

5 Conclusions

Australia is the second largest global exporter of beef in the world after Brazil. The Australian beef industry maintains a strong emphasis on producing beef from sustainable production systems, predominantly from the extensive rangeland areas of eastern Australia. While a number of studies have been conducted to quantify the resource use and environmental impacts of Australian beef production, none have investigated the impacts of producing and exporting premium beef to the USA. Australia supplies both premium grass-fed and grain-fed beef to the USA, much of which is destined for consumption in the food services sector. This study investigated the resource use and environmental impacts associated with producing, processing and exporting this beef to the USA. Results of GHG emissions from the present study were of a similar order to previous Australian LCA results when compared at the farm gate stage. Greenhouse gas emissions were similar to those previously reported in the Australian literature, while water use was higher. Few energy use or land occupation results have been reported previously for Australian beef. GHG emissions and total land occupation were lower for grain finished beef than for grass finished beef, while energy use tended to be higher.

Beef is a globally traded product, and concerns may exist regarding the impacts of transport on the environmental sustainability of beef. This study found that transport had a modest impact (<4%) on GHG emissions, suggesting transportation distance is not a suitable indicator of this impact. Greenhouse gas emissions (excl. LU and dLUC) tended to be lower than reported for many Northern Hemisphere countries. From review of the contribution analysis of these studies, the contribution of nitrous oxide and carbon dioxide emissions was lower in the present study than most studies from the northern hemisphere. When emissions from LU and dLUC were included, emissions increased 6-34% but remained in the range of values reported for Northern Hemisphere countries, many of which did not report LU and dLUC emissions. Emissions from LU and dLUC are expected to decline to very low levels over the next 12 years in response to policy changes in Australia. These results should be reviewed at three-five year intervals to observe this decline in emissions.

Energy use from the Australian production systems tended to be lower than most studies from the Northern Hemisphere despite the increased transport distances which contributed to total energy use, while land occupation was higher on average. Few data were available for water use, though it appears that water use is likely to be lower than in the USA from the two studies available because of the lower reliance on irrigation in Australia.

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Appendix 1

Uncertainty

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

Farm Inventory Data

Farms use a range of inputs including energy for transport and farm operations, inputs for crop and pasture production (fertilisers, chemicals), and inputs associated with livestock (veterinary products, feed). Additionally, farms relied on a number of services such as accounting, banking and communications.

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

In order to improve comparability between farms, the farm inventory data are presented here (Table 9, Table 10) per tonne of DMI consumed. Feed intake is a common unit for considering the stocking capacity of a farm and is a reasonable comparative unit. These values can be converted to dry sheep equivalents (DSE) using a value of ~400 kg DMI per DSE, or can be converted to cattle Adult Equivalents (typically a 450 kg steer) using an approximate annual feed intake value of 2.6 t DMI.

Table 9 – Material inputs for QLD and NSW case study farms (n = farm numbers)

			NSW (n = 5)	QLD (n = 5)	
Inputs	**Data source description**	**Units**	**per tonne DMI**	**per tonne DMI**	**Uncertainty (SD or range)**
Feed	Data collected from farm				
Dry lick		kg	0.13	5.86	1.06
Hay		kg	1.11	2.81	1.06
Cereal grain		kg	25.11	0.00	1.06
White fluffy cotton seed		kg	0.00	0.002	1.06
Energy	Data collected from farm				
Electricity		kWh	5.94	1.24	1.01
Oil		L	0.08	0.06	0.04-0.09
Diesel		L	0.88	2.74	1.01
Petrol		L	0.76	0.13	0.09-0.84
Fertilisers	Data collected from farm				
Superphosphate		kg	5.98	0.00	0-9.4
Urea or compound fertiliser			1.26	0.00	1.01
Pesticides	Data collected from farm	g	10.63	0.00	1.04
Other inputs and services	Data collected from farm				
Veterinary services		$	0.82	1.93	1.92
Communication services		$	0.66	0.46	1.92
Insurance		$	1.94	1.02	1.92
Accounting		$	1.00	0.67	1.92
Industry levy		$	0.67	1.04	1.92
Contract helicopter mustering		$	0.00	0.43	1.92

Table 10 – Material inputs for QLD and NSW regional average farms (n = farm numbers)

			NSW (n = 51)	**QLD (n = 64)**	
Inputs	**Data source description**	**Units**	**per tonne DMI**	**per tonne DMI**	**Uncertainty (SD)**
Feed	ABARES				
Dry lick		kg	0.76	6.17	1.07
Hay		kg	19.12	22.22	1.07
Grain		kg	4.55	4.94	1.07
Energy	ABARES				
Electricity		kWh	3.02	2.10	1.07
Oil		L	0.17	0.05	1.12
Diesel		L	4.58	3.82	1.01
Petrol		L	0.71	0.44	1.01
Fertilisers	ABARES				
Superphosphate		kg	2.93	0.00	1.06
Pesticides	ABARES	g	694.66	150.43	1.06
Other inputs and services	ABARES				
Veterinary services		$	3.43	2.18	1.33
Communication services		$	0.55	0.51	1.33
Insurance		$	1.58	1.04	1.33
Accounting		$	0.86	0.52	1.33
Industry levy		$	0.39	0.37	1.33

Feedlot inventory data

Feedlot inventory data were collected over a three year period from detailed metering and monitoring of energy use, water use, commodity use and livestock numbers and performance (full details are available from Davis et al. 2008a, 2008b). Manure production was estimated from feed and cattle performance data using the BeefBal model, and additional input data were collected from the feedlot managers as required. Financial records were confidential for all of the feedlots and were not available. These data were estimated from one feedlot where such data were provided and were allocated on a "per head day" basis across the feedlots (Table 11, Table 12).

Table 11 – Material inputs and outputs for the MF feedlot

Inputs	Data source description	Units	Per animal finished (421 - 622 kg)	Uncertainty (SD or range)
Cattle	Data collected from feedlot	kg	421.0	
Feed ration	Data collected from feedlot	kg DM	1416.1	1.06
Land occupation	Data collected from feedlot			
Non arable (Feedlot)		m^2	12.6	1.20
Arable (effluent irrigation area)		m^2	12.9	1.20
Energy	Data collected from feedlot			
Electricity		kWh	3.2	1.01
Diesel		L	1.5	1.01
Petrol		L	1.2	1.01
Transport	Estimated transport distances for cattle and feedlot commodities	t.km	84.4	
Other inputs and services				
	Veterinary services	$	17.0	1.92
	Communication services	$	0.3	1.92
	Accounting	$	11.9	1.92
	Industry levy	$	5.1	1.92
	Horse feed	kg	0.6	1.92
	Staff travel	km	1.0	1.92
Outputs				
Finished animal	Animal to abattoir	kg	622.0	
Excreted Manure				
Manure N	Mass Balance	kg	27.9	±10%
Manure VS	Mass Balance	kg	317.6	±10%
Manure P	Mass Balance	kg	3.5	±10%
Manure K	Mass Balance	kg	8.8	±10%
Emissions				
Enteric methane	Modelled from feed data using Moe & Tyrell (1979) and Beauchemin et al. (2008)	kg	16.4	

Table 12 – Material inputs and outputs for the LF feedlot

Inputs	Data source description	Units	Per animal finished (442 - 760 kg)	Uncertainty (SD or range)
Cattle	Data collected from feedlot	kg	442.0	
Feed ration	Data collected from feedlot	kg DM	3686.5	1.06
Land occupation	Data collected from feedlot			
Non arable (feedlot)		m^2	25.5	1.20
Arable (effluent irrigation area)		m^2	70.4	1.20
Energy	Data collected from feedlot			
Electricity		kWh	10.1	1.01
Diesel		L	11.3	1.01
Petrol		L	1.1	1.01
Transport	Estimated transport distances for cattle and feedlot commodities	t.km	88.7	
Other Purchases and inputs (expenses)				
	Veterinary services	$	49.6	1.92
	Communication services	$	0.9	1.92
	Accounting	$	34.5	1.92
	Industry levy	$	5.1	1.92
	Horse feed	kg	1.8	1.92
	Staff travel	km	3.0	1.92
Outputs				
Finished animal	Animal to abattoir	kg	760.8	
Excreted Manure				
Manure N	Mass Balance	kg	75.9	±10%
Manure VS	Mass Balance	kg	772.9	±10%
Manure P	Mass Balance	kg	11.3	±10%
Manure K	Mass Balance	kg	41.6	±10%
Emissions				
Enteric methane	Modelled from feed data using Moe & Tyrell (1979).	kg	65.6	

Feed milling and rations

Feed milling inventory data for all feedlots were based on records kept by the feed mills. These data are presented in Table 13.

Table 13 – Major inputs for feed milling from Australian feed mills

Inputs	Data source description	Units	MF feedlot (per tonne delivered to bunk)	LF feedlot (per tonne delivered to bunk)
Energy				
Electricity	Data collected from feedlot	kWh	6.8	6.4
LPG		L	0.4	n.a.
Butane		m^3	n.a.	1.7
Diesel		L	11.6	2.4
Water	Data collected from feedlot	L	120.2	96.8
Transport	Est. transport distances for commodities to the feedlot	t.km	177	175

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

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

Feed data were collected for total feed intake over three years. Commodity inputs for the cattle rations were obtained from the feed mill and from the feedlot nutritionist. There are many rations fed throughout the year with a different formulation based on the nutritional requirements of the animals and the cost of inputs. Animal inputs averaged across the rations are show in Table 14.

Table 14 – Aggregated, simplified rations for the MF and LF feedlot

Commodities (protein content in brackets)	Amount MF	Amount LF
	kg DM	*kg DM*
Barley (10%)	100.2	40.4
Maize (8%)		35.0
Wheat (13%)	512.7	410.0
White fluffy cottonseed	121.9	
Lucerne Hay	50.9	
Wheat Hay	3.7	
Cotton Hulls	72.4	
Tallow	37.7	
Feed additives	100.6	23.6
Sunflower (36%)		2.9
Cereal straw		132.0
Maize Silage		122.5
Pasture Silage		23.2
Wheat Silage		16.5
DDG (dried distillers grain)		57.6
Mill Mix		57.3
Molasses		79.5
Total	**1000.0**	**1000.0**

Meat processing data

Inputs and impacts associated with meat processing were collected from two meat processing plants and from an industry survey of beef processing plants (GHD 2011). These input data are shown in Table 15. Emissions of refrigerants were not available for meat processing because of a lack of data.

Table 15 – Major inputs associated with meat processing

Major Inputs	units	Per tonne carcase weight
Water use, 100% consumptive	L	8743.3
Energy Use		
Electricity	kWh	318
LPG	MJ	83.47
Diesel	MJ	39.92
Petrol	MJ	7.26
Coal	MJ	693
Natural Gas	MJ	1230

Total impacts from meat processing (per tonne carcase weight processed) were 0.80 kg CO_2-e, 5.8 MJ energy use, and 8.75 L water. Greenhouse gas emissions from effluent were included. These values differ from the contribution analyses in the results section because they are presented with a different functional unit in the results, and because in the results the avoided products associated with co-products are grouped with meat processing, lowering the total impacts.

Transport and warehousing

Transport – Australian processor to USA warehouse

Transport from the meat processing plant to the port was estimated from a weighted average of six meat processing plants in the region to the port of Brisbane, providing an average total transport distance of 450 km. Transport was via B-Double (articulated) trucks with a load capacity of 38 t.

Import data from the US Trade Census[1] shows that the major ports of entry for Australian products are Los Angeles and Philadelphia. This study assumed imports were received to the port of Los Angeles (shipping distance of 11,921 km) or Philadelphia (shipping distance of 18,117 km). Energy and GHG emissions during refrigerated shipping was taken from Webb et al. (2013).

Energy and GHG emissions during refrigerated shipping was taken from Webb et al. (2013).

Products arrive at port in containers which are taken directly to a facility to clear customs and USDA inspection. This is frequently a large warehouse located within a few kilometres (~30 km) of port. The container is transported with a specialised truck to the warehouse (drayage). At this facility the customs seal from the container is broken, the product unloaded, inspected and then stored until delivery to the importer. Depending on a number of factors, the meat (particularly frozen product) may be kept at this facility between 30 and 90 days. In this study, it was assumed that chilled product was stored for less than 30 days. One importer indicated that the hold time could be as short as 2 weeks, but this was not typical. After warehousing, product is shipped throughout the country. For the purposes of the study, 50% of product was assumed to be transported an average of 200 km, and 50% was assumed to be transported an average of 1000 km. These products are shipped in diesel powered long-haul combination trucks.

Refrigerated warehouse storage

The impacts associated with storage in refrigerated warehouse were estimated in two ways. Micro data from the Energy Information Agency Commercial Buildings Energy Consumption Survey[2] were used to estimate the energy use associated with warehouse storage. Based on EIA survey data, refrigerated warehouses consume, on average, 30.44 kWh / (m^3 yr) and natural gas consumption of 21,019 BTU / (m^3 yr) based on an estimated 9m typical warehouse height and an 80% utilization rate that accounts for aisles and other overhead floor space. Based on ASHRAE design guidelines, the energy consumption for electricity is 33.55 kWh / (m^3 yr) and for natural gas, 35,030 BTU / (m^3 yr). These data were cross checked by surveying meat industry warehouse managers. One plant manager reported in an interview that electricity consumption at his facility was on the order of 5 kWh / (m^3 yr), inclusive of dock staging and electric forklift operation. Thus there is an approximately 6-fold range in estimated electricity demand in estimates – individual plants reporting in the EIA survey range from 5 to over 60 kwh / (m^3 yr). In lieu of a specific data, the study utilised average energy use values from the EIA survey which were considered conservative based on interviews with plant managers.

[1] https://usatrade.census.gov/

[2] http://www.eia.gov/consumption/commercial/data/2003/index.cfm?view=consumption#c1

Unit processes for transportation

The US lifecycle inventory published by the national renewable energy laboratory, and available from the USDA digital commons has approximately 100 distinct transportation data sets. For the US supply chain of Australian meat products, there are two distinct transport steps, as described above. The first is a short haul diesel truck moving the container from the ship yard to the USDA inspection and initial warehouse location. The second stage begins when the importer collects the product from the warehouse for distribution to their customers. In general, there is no additional intermediate storage by the importer. The US LCI data sets chosen for the short haul drayage was the combination truck, for which your processes on the West Coast in the north-eastern United States have been created based on the US EPA MOVES 2010a and Argonne National Laboratories Greet models. For the long-haul transport, while there were several distinct models in the US LCI data set, though the differences between each were small.

Background data sources

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

Appendix 2 – Water use inventory

Methodology

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

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

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

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

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

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

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

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

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

An additional component of the inventory is the relationship between Land occupation and water availability. When assessing the impact of an agricultural system, it is important to identify whether the system alters the flow of runoff to the environment as this is a component of water use. Milà i Canals et al. (2009) proposes a method whereby the difference in evapotranspiration between the system investigated and a reference system (i.e. natural vegetation) is used to determine the effect of the system on the water balance. Where a system evapo-transpires more water than the reference system, this results in additional water use that is attributable to the product grown on that land. Likewise, if a production system utilised less water than the reference system (as is often the case in Australia) a negative flow (or credit) may be applied. This important aspect has not been thoroughly investigated here, indicative values are provided.

Data collection and modelling approach

The water inventory was developed by using a series of water balances for important processes in the foreground system. Full characterisation of water sources (inputs) and outputs from each stage were determined, including all losses.

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

Foreground system for farms:

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

Foreground system for feedlots:

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

Background system for farms and feedlots:

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

Consumptive water use data for background processes are not well documented within the AustLCI and Ecoinvent databases. Water use within background databases tends to be 'input water' only; consumptive and non-consumptive uses are not differentiated. Background water use was reviewed to determine important processes (i.e. processes contributing >1%) and these processes were standardised to the methods used here where required. Methods and assumptions used to determine water use in each stage are provided in the following sections.

Farm water inventory

Modelling livestock drinking water use

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

Several factors determine drinking water intake for cattle, including feed intake, ambient and water temperature, class of animals and live weight (National Research Council 1996). Water use can be particularly variable in response to climate. The drinking water prediction equations from Ridoutt et al. (2012b) were applied in the present study. The feedlots under investigation in this study had metered records of water use from a previous study (Davis et al. 2008a, b). Table 16 provides climate data relevant to the farm water modeling.

Table 16 – Summary of site data used in water modelling for the case study farms and feedlots

Region	Regional average Rainfall (mm / yr)	Regional average Evaporation (mm / yr)	WSI
Central/Southern QLD	674	1972	0.021 / 0.85
Nth NSW	733	1573	0.021
Short-fed Feedlot	819	1717	0.021
Mid-fed Feedlot	524	1584	0.021
Long-fed Feedlot	819	1379	0.021

Water sources

An assessment of the water supply was made at each farm, based on records and input from the farmers and from an analysis of the property layout. Based on this analysis, the breakdown of water sources for the case study farms was determined (Table 17).

Table 17 – Sources of water supply for farms and feedlots

Source of water supply	QLD CSF	NSW CSF	MF Feedlot	LF Feedlot
	% of total water supply			
Dam	45%	65%	28%	0%
Creek/River	16%	20%	0%	100%
Bore	39%	15%	72%	0%

Direct supply from creeks and rivers

Supply losses associated with direct extraction were negligible because there was no supply network. Evaporation from river and creek water surfaces was excluded, as this was part of the natural system and therefore not attributable to livestock production.

Farm dams

Losses associated with water supply from farm dams (Figure 10) were modelled using farm dam water balances constructed from long term (30 year) climate data for each farm. Dams and catchment areas were assessed during site visits and were later mapped using aerial imagery. Catchment runoff (dam inflow) was modelled using USDA-SCS KII curve numbers (USDA NRCS 2007), with appropriate values determined from site observations of soil type and farming practices. Runoff predictions were calibrated at the local scale using farmer knowledge of the frequency of runoff events, and against catchment yields for similar catchments. Dam volumes were modelled from top water level surface area measurements taken in GIS, on-site assessments at each farm and evaluation by the farm owners. Farm dam water balances were constructed from the average volume, extraction rates and catchment size for each farm. The dam water balances were modelled using a daily time-step water balance using long term rainfall and evaporation data obtained for each region as Patched Point Datasets from the SILO database (DSITIA 2013, Jeffrey et al. 2001). The balance accounted for extractions, seepage and evaporation losses. Seepage losses were only noticeable from poorly constructed dams and were assumed to be negligible for the majority of dams. In cases were dam seepage was evident, this was typically through the wall of the dam, resulting in soak areas below the dam wall. Water seepage was considered a consumptive use, because this water was eventually lost via evapo-transpiration below the dam. The dam water balances were calibrated using records of filling and emptying events for each region, determined through discussion with the farmers. Evaporation was predicted from pan evaporation after applying a pan factor, varying from 0.75-0.9, which were similar to values suggested by Burman & Pochop (1994) and Craig (2006).

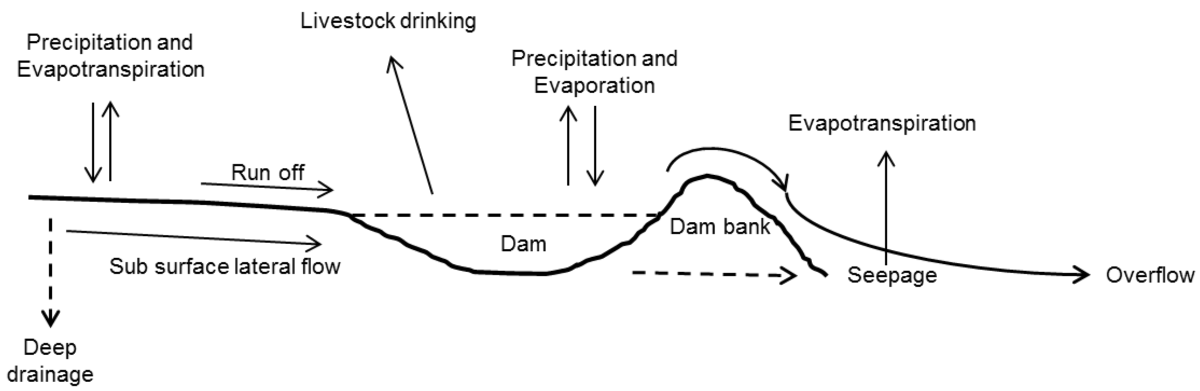

Figure 10 – Illustration of farm dam water supply system modelled in the study

We developed a ratio to describe the total water intercepted from the environment proportional to the water extracted for livestock, referred to as the intercept to extraction ratio (Table 18). The intercept to extraction ratio is based on the volume of water intercepted from the environment as a result of dam construction (the difference between catchment runoff volume and overflow volume) relative to the volume of water extracted for livestock drinking. Higher ratio values indicate a greater volume of water intercepted from the environment to provide water for livestock.

Table 18 – Dam supply efficiency factors

Inflows	QLD CSF	NSW CSF	MF Feedlot	LF Feedlot	Uncertainty (SD)
Intercept to utilisation ratio	4.49	4.32	3.0	n.a.	1.45

Intercept to utilisation ratios were influenced primarily by net evaporation rates, dam density and dam construction. Net evaporation was higher for the Queensland farms, but dam density was lower and dams were deeper on average. Consequently, the supply efficiency was similar.

Reticulated supply from bores or rivers

Bore and reticulated river water use was not metered at any of the farms, and loss rates were estimated based on a review of the water system and discussions with the farmers. The greatest losses were associated with leaks and overflowing tanks or troughs, and evaporation from open tanks and troughs. Losses ranged from 5-13% of total extraction.

Feedlot water inventory

Feedlot water use data were available from the detailed study conducted by Davis et al. (2008a). This study measured water use at seven feedlots over two years in detail. Based on these data for total water use, we modelled consumptive water uses throughout the feedlot. All drinking water was considered a consumptive use, though in some cases the evaporative loss is a secondary process, such as the evaporation of urine or moisture from manure at the feedlot or after land application. We based prediction of total consumptive water use on the final fate of water as a result of the production system.

Additional water use activities consisted of cleaning and minor water uses. Cleaning water use was made up of cattle washing, while minor uses included the trough cleaning water, evaporation from the troughs, and office and amenities water usage.

Feedlot controlled drainage area water balance

Australian feedlots are designed to control drainage and overland flow around the feedlot site to restrict movement of manure nutrients to the environment. Within the controlled drainage area, runoff is greatly increased from hard surface areas (pens, roads). All water is captured in engineered effluent ponds, which are constructed with a storage capacity to limit effluent release to a one in ten year rainfall event. Excess water from the feedlot controlled drainage area either evaporates from the effluent pond, or is irrigated to grow crops. Because the feedlot site is highly modified, the most accurate way to determine the impact of the feedlot on the local hydrology was to compare the site to a reference, or ‘green field’ site (i.e. the feedlot site in the absence of the feedlot). We did this by modelling runoff from the greenfield site using USDA-SCS KII curve numbers (USDA-SCS 1972, USDA NRCS 2007). For the purposes of the water balance, we assumed the feedlot site released no water. Runoff from the reference, green field site was attributed to the feedlot as a water use. Data are shown in Table 19.

Table 19 – Runoff from reference land occupation attributed to feedlot cattle production at two feedlots

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

Appendix 3 – Modelling GHG emissions

Livestock emissions

Methods and factors

Feed intake for grazing cattle was modelled from the livestock inventory and livestock growth rates reported for each farm using the feed intake equation from Minson and McDonald (1987) as applied in the Australian NGGI. For feedlot cattle, feed intake was based on feed intake records from the feedlots surveyed over a two year period.

The parameters and equations used in this study to determine the GHG emissions from grazing and feedlot beef are summarised in Table 20 and Table 21, along with the assumed uncertainty. Nitrogen leaching from fertiliser and animal manure was only included when the ratio of evapotranspiration to annual precipitation is lower than 0.8 or higher than 1 (DCCEE 2010).

Table 20 – GHG parameters used for grazing cattle with uncertainty

Emission source	Key parameters / model	Assumed Uncertainty	Reference
Enteric methane (temperate climate)	M (kg/hd) = (Y (% Gross Energy Intake as CH_4) / 100) x (GEI (MJ/kg) / F (MJ / kg CH_4)	± 20%	DCCEE (2010) – from Blaxter and Clapperton (1965)
Enteric methane (tropical climate)	19.6 g CH_4 / kg DMI x kg DMI / hd	± 20%	Kennedy and Charmley (2012)
Manure methane	M (kg/hd) = I (kg DM/hd) x (1 - DMD) x MEF	± 20%	DCCEE (2010)
Manure nitrous oxide	0.004 kg N_2O-N / kg N in urine 0.005 kg N_2O-N / kg N in faeces	± 50%	DCCEE (2010)
Manure ammonia	0.2 kg NH_3-N / kg N of excreted in manure	± 20%	DCCEE (2010)
Indirect nitrous oxide from ammonia losses	0.01 kg N_2O-N / kg NH_3-N volatilized	± 50%	DCCEE (2010)
Indirect nitrous oxide from leaching and runoff	0.0125 kg N_2O-N / kg NO_3-N lost in leaching and runoff	± 50%	DCCEE (2010)

Table 21 – GHG parameters used for feedlot cattle with uncertainty

Emission source	Key parameters / model	Assumed Uncertainty	Reference
Enteric methane	M (kg/hd) = (3.406 + 0.510SR + 1.736H + 2.648C) / F (MJ / kg CH_4)	± 20%	DCCEE (2010) – from Moe and Tyrrell (1979)
Manure methane	M (kg/hd) = VS (kg/head) x B_o (0.17 m^3 CH_4/kg VS) x MCF x p (0.622 kg/m^3)	± 20%	DCCEE (2010)
Manure nitrous oxide	Faecal and urinary N – 0.005 kg N_2O-N / kg N in faeces.	± 50%	Muir (2011)
Manure ammonia	0.75 kg NH_3-N / kg N of excreted in manure	± 20%	Watts et al. (2012)
Indirect nitrous oxide from ammonia losses	0.01 kg N_2O-N / kg NH_3-N volatilized	± 50%	DCCEE (2010)

Feedlot feed parameters

Table 22 – Daily feed intake and feed properties for two feedlot rations

	Units	Mid-fed feedlot (110-150 DOF)	Long-fed feedlot (300+ DOF)
Daily Intake (assume DMI)	(kg/day)	10.6	8.56
Proportion of grains in feed		0.845	0.684
Proportion of concentrates in feed		0.100	0.022
Proportion of grasses in feed		0.004	0.294
Proportion of legumes in feed		0.051	0.001
Proportion of oil in feed		0.04	0.001
Enteric methane production – without accounting for oil	kg/hd/d	0.183	0.195
Dietary CP	%	15.9%	17.2%

Appendix 4 – Land use change GHG methods and data

Direct land use change – grazing land

Carbon dioxide emissions from clearing for beef were calculated for each relevant bioregion in QLD and CMA in NSW using the following formula:

CO_2-e_clearing = (1.25 * M* A_rem + 1.25 * 0.2 * M * A_rg) * 0.5 * 3.67

Where:

- M is the average maximum biomass for each bioregion and CMA (from 30 random points in each)
- 1.25 is a constant included to transform maximum biomass estimates, which are for above ground plant parts, to an estimate of whole tree biomass
- A_rem is the area of remnant woody for pasture for each bioregion and CMA. This was estimated from the extent of woody clearing reported by SLATS for pastoral purposes, multiplied by a factor for the proportion of clearing that was remnant woody clearing (see below for details). While QLD SLATS reports provide extent of clearing for the purpose of pasture, NSW SLATs reporting estimates extent cleared at a higher level for "crop, pasture, thinning". For CMAs, clearing for pasture was estimated as 95% of clearing reported for "crop, pasture, thinning". The 95% figure is based on estimates from similar regions in QLD. An average of 92% of total clearing (not just agricultural clearing) since 1990 was for pasture in QLD's Brigalow Belt bioregion, and the same figure for QLD's New England Tablelands bioregion was 95%. Southeast Queensland had a much lower proportion of total clearing attributable to pasture but other major clearing purposes in SEQ (forestry and housing) are also distinguished in the NSW SLATs analysis.
- A_rg is the area of non-remnant woody clearing for pasture for each bioregion. Like A_rem, A_rg was estimated by multiplying area cleared for pastoral purposes by the percentage of clearing that affected non-remnant woody vegetation (see below for details).

The relative extent of woody clearing that affects remnant and non-remnant vegetation (A_rem and A_rg from total clearing) for realistic estimates of biomass. Maximum biomass was based on National Inventory Reporting methodology accessed through FullCAM (DCCEE 2013).

Annual sequestration was estimated as 1.5% of the maximum biomass. This is an average rate for approximately the first 30 years of regrowth (i.e. 30 year old regrowth is expected to have ~45% of maximum biomass and 45/30 = 1.5).

Inventory results are presented in Table 23

.

Table 23 – Estimated GHG emissions from land clearing for beef production regions in southern-central Queensland and northern New South Wales showing:

	Beef production regions	
	Central-Southern QLD	Northern NSW
LUC GHG emissions inclu. 1990-2010 (t CO_2-e/ha/yr)	0.46	0.05
LUC GHG emissions Projected from 2026 (t CO_2-e/ha/yr)	0.14	0.05
C sequestration in woody regrowth (t CO_2-e/ha/yr) Low scenario	0.03	0.02
C sequestration in woody regrowth (t CO_2-e/ha/yr) High scenario	0.21	0.17
Net LUC GHG emissions inclu. 1990-2010 (t CO_2-e/ha/yr) Low scenario	0.25	-0.12
Net LUC GHG emissions inclu. 1990-2010 (t CO_2-e/ha/yr) High scenario	0.43	0.03
Net LUC GHG emissions Projected from 2026 (t CO_2-e/ha/yr) Low scenario	-0.07	-0.12
Net LUC GHG emissions Projected from 2026 (t CO_2-e/ha/yr) High scenario	0.11	0.03

Table 24 – LUC data and derived GHG fluxes for the Central-southern Queensland and northern New south Wales beef production regions. Analysis was based on three Queensland bioregions and eight New South Wales CMAs to correspond to the ABARES regions used in the cattle analysis and to the best available satellite imagery data.

	Brigalow Belt	**New England Tableland**	**Southeast Queensland**	**Border Rivers-Gwydir**	**Central West**	**Hawkesbury-Nepean**	**Hunter-Central Rivers**	**Lachlan**	**Namoi**	**Northern Rivers**	**Southern Rivers**
Region area ('000s ha)	36391	775	6174	4751	8709	2131	3328	8140	4834	5058	3012
Avg max biomass (t/ha)	85.9	95.6	200.3	117.5	105.6	160.1	154.6	79.7	122.2	252.3	221.1
Standard error for avg. max biomass	8.2	7.5	20	8.6	9.5	13	19	8.3	17.9	30	34.6
Clearing rate 1990-2010 ('000s ha/yr)	164	2.38	6.98	1.8	3.54	0.27	1.05	2.27	1.05	2.12	0.34
Clearing rate 2006-2010 ('000s ha/yr)	52.2	1.86	5.98	2.54	2.08	0.23	0.97	1.52	0.98	3.02	0.24
Clearing % regrowth 1990-2010	56	70	70	56	56	70	70	56	56	70	70
Clearing % regrowth 2006-2010	68	81	77	56	56	70	70	56	56	70	70
Clearing emissions 1990-2010 (Mt CO2-e)	370.56	4.56	27.37	4.91	10.26	0.89	3.32	4.89	3.29	9.86	1.59
Clearing emissions 2006-2010 (Mt CO2-e)	19.16	0.59	4.31	1.51	1.11	0.15	0.61	0.61	0.6	3.08	0.21
Regional rate of clearing emissions 1990-2010 (tCO2-e/ha/yr)	0.509	0.294	0.222	0.052	0.059	0.021	0.05	0.03	0.034	0.097	0.026
Regional rate of clearing emissions 2006-2010 (tCO2-e/ha/yr)	0.026	0.038	0.035	0.08	0.032	0.017	0.046	0.019	0.031	0.152	0.017
Estimated regrowth area on pastoral land ('000s ha)	2346	114	634	473	512	19	143	349	215	189	17
Estimated range for sequestration 1990-2010 (Mt CO2-e)	36.96-110.89	2.01-6.02	23.29-69.88	5.56-16.68	5.41-16.23	0.3-0.91	2.21-6.62	2.78-8.34	2.62-7.86	4.77-14.32	0.37-1.12
Estimated range for regional sequestration rate (tCO2-e/ha/yr)	0.05-0.15	0.13-0.39	0.19-0.57	0.06-0.18	0.03-0.09	0.01-0.02	0.03-0.1	0.02-0.05	0.03-0.08	0.05-0.14	0.01-0.02

Land use and direct land use change – cultivated land

Crop land LU and dLUC emissions were determined from state wide cropping areas using data from ABS and assumptions from the DCCEE (2012b). The expansion of crop land was determined by comparison of the largest area of land cultivated for cereal crops prior to 1990 with the largest area cultivated in the five years to 2010, as reported by ABS (2013). This analysis revealed a 25% expansion of crop land in QLD and a 12% expansion in NSW. Expansion of crop land in Australia has predominantly been from grassland (DCCEE 2012b), assumed to be 70% and 80% respectively for QLD and NSW, with the remaining area being from forest. Total carbon losses were assumed to be 12.6 t C / ha for conversion of grassland to crop land, and 84 t C / ha for the conversion of forest land to crop land based on tier II methods (DCCEE 2012b). This resulted in annualised emission rates of 15.5 and 2.3 t CO_2 / ha.yr for land converted from forest and grassland respectively.

Soil carbon losses are known to continue for longer than 20 years after an LUC event, with the majority of losses occurring within 40 years for most soils (Dalal & Mayer 1986). To determine additional C losses expected to occur from soils where land conversion occurred before 1990, we determined the land converted in the prior 20 years using the same method based on national crop land statistics for each state (ABS 2013). This analysis revealed an additional 31% of QLD crop land and 10% of NSW crop land was converted to cropping in the period 1970-1990 and soil carbon losses were included assuming losses of 0.6 t C / ha.yr (Dalal & Mayer 1986). Soils converted to cropping prior to 1970 were assumed to have reached a new steady state soil C level.

Attribution of emissions associated with expanded grain production requires an understanding of the factors contributing to an increase in grain production. Australian grain production has increased following a strong, linear trend since 1960 (analysis of data from ABS (2013) – see Figure 11). The increase in grain production is partly a response to increased crop yields which rose from an average of 1.2 t / ha in the ten years to 1970, to an average of 1.7 t / ha in the 10 years to 2010 (ABS 2013). The remaining increase in crop yield has been in response to conversion of land to cropping. Conversion of land for grain production is primarily an economic decision based on relative returns from grain or alternative uses such as grazing. Australian grain prices are closely linked to international prices rather than local grain users and there is relatively little specialist feed grain production in Australia. Considering this, it is difficult to determine a causal association between livestock grain use and grain production. In the current project, we assumed that the feedlot industry contributed equally to the expanding demand for grain production and the attributed impact was therefore proportional to the overall rate of expansion in each state.

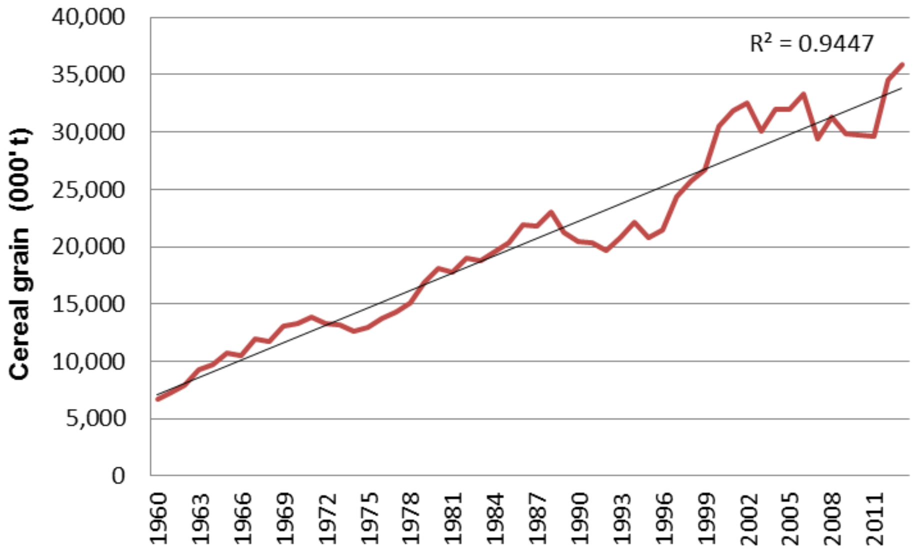

Figure 11 – Five year rolling average cereal grain production in Australia from 1960 to 2012 (data accessed from ABS 2013)

Emissions from LU and dLUC associated with cultivated land and subsequently grain production, based on the method outlined, were 1.75 t CO_2-e / ha.yr and 0.8 t CO_2-e / ha.yr for QLD and NSW respectively.