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The Environmental sustainability of Australian lamb 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 lamb production in three regions of south 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. Energy demand ranged from 19.1 to 28.5 MJ / kg lamb. Consumptive water use ranged from 125.2 to 481.1 L / kg lamb, while stress-weighted water use ranged from 12.1 to 280.9 L H_2O-e / kg lamb. Occupation of cultivated arable land averaged 2.1 m^2 / kg lamb. Total greenhouse gas (GHG) emissions ranged from 13.4 to 16.0 kg CO_2-e / kg lamb. The farm production system contributed the majority of impacts across all categories, with transport and storage collectively contributing ≤ 6% to all impact categories with the exception of energy use, where the contribution was higher. These results suggest that the impacts from the transport of lamb, even where these distances are large, are not a significant driver of the environmental impact of this product.

Executive summary

Australia is the second largest global exporter of sheep meat in the world after New Zealand. The Australian sheep industry maintains a strong emphasis on producing lamb from sustainable production systems, predominantly from the rangeland areas of south eastern Australia. However, to date few studies have been done to quantify the resource use and environmental impacts of producing Australian lamb for major markets such as the USA. 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 lamb from three major production regions in south eastern Australia to the USA. The study followed a Life Cycle Assessment (LCA) method for the production of one kilogram of retail ready lamb cuts in the USA. The specified end point was the distribution warehouse located in the eastern United States.

Energy demand ranged from 19.1 to 28.5 MJ / kg lamb. Consumptive water use ranged from 125.2 to 481.1 L / kg lamb, while stress-weighted water use ranged from 12.1 to 280.9 L H_2O-e / kg lamb. Occupation of cultivated arable land averaged 2.1 m^2 / kg lamb. Total greenhouse gas (GHG) emissions ranged from 13.4 to 16.0 kg CO_2-e / kg lamb. The GHG emissions associated with land use and land use change were found to be either a small emission source (0.3 kg CO_2 eq/kg lamb), where soil carbon sequestration was assumed to be zero, or modest removals ranging from -4.0 to -0.8 kg CO_2 eq/kg lamb when the soil carbon sequestration under fertilised pastures was included in the assessment.

The primary production stage of the supply chain contributed the largest impacts for all impact categories, ranging from 28-52% (energy demand) to ≥90% for GHG, consumptive water and land occupation. As a result, variation between supply chains mainly related to regional differences such as climatic variability, variation in water resources and scarcity, and variation in management.

Lamb is a globally traded product, and concerns may exist regarding the impacts of transport on the environmental impact of lamb. The study confirmed findings from previous studies in the literature by showing that GHG impacts from the transport of Australian lamb to the USA are relatively minor (6%) despite very long travel distances. This confirms that transport distance or ‘food miles’ is not a suitable indicator of the emissions intensity of Australian lamb exports.

This research applied a multi-tier data collection approach which included detailed analysis of a small number of case study lamb production farms and application of a much larger regional survey dataset. This focus on detailed assessment of the production phase aligned with the critical impact areas in the supply chain, improving the robustness of the study.

This study presents the first multi-impact study of its kind for Australian export lamb. The study has applied new datasets, new analysis methods and new methods for handling the co-production of wool, live weight, and co-products from meat processing. These methods are considered more robust and are more closely aligned with the ISO standards for LCA.

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1 Introduction

1.1 Background

Australia has been the second largest exporter of sheep meat in the world for several years, closely following New Zealand in total volume exported annually (FAO 2011). Australian lamb is exported to many countries and regions, the largest of these being Middle East, China, the USA, PNG and the European Union. In the USA and around the world, retailers and consumers are seeking information regarding the provenance and environmental performance 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 environmental impacts 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 environmental impacts 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 emissions were only 5% despite the very long transport distance (~17,000 km). These results suggest that a more holistic measure of the environmental impacts 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.

To date, there has been no holistic analysis of the environmental impacts of Australian lamb production, processing and transport to markets in the USA. The present study provides such an analysis and 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 (resources and energy) and outputs (product, emissions and waste) to the environment are quantified for each process and then combined in the process flow chart. Allocation of inputs and outputs needs to be clarified where processes have several functions (for example, where one production system produces several products). In this case, different process inputs and outputs are attributed to the different goods and services produced. An extra simplification used by LCA is that processes are generally described without regard to their specific location and time of operation.

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

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

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

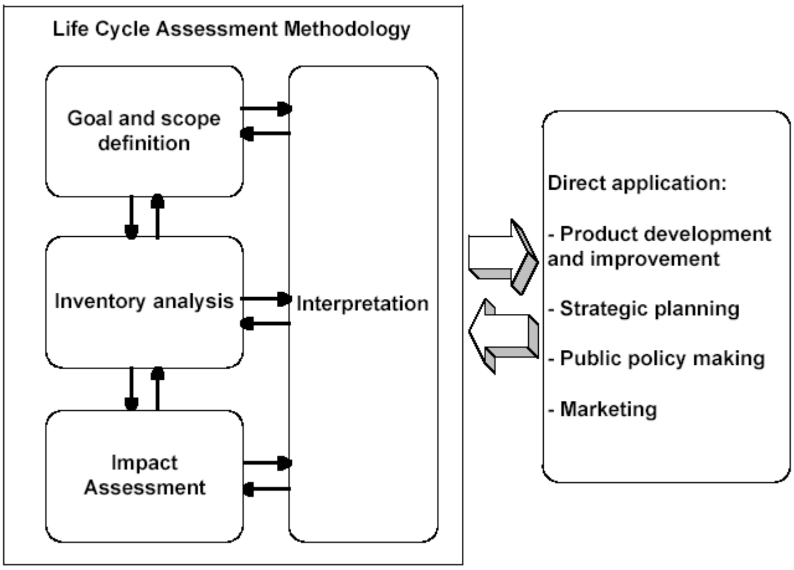


Figure 1 – General Framework for LCA and its Application (ISO 2006a: 14040)

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

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

It is common for a single product (such as lamb) to involve several thousand 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 over which the study team have access and where data are directly collected. This 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 prime lamb 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 options for handling co-production according to ISO 14044 (ISO 2006a) in order of preference are:

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

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

1.3 Sustainability of Australian lamb production

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 provision of clean air and water. Producing lamb in a sustainable production system is a high priority for the Australian sheep 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 sheep 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 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.

The focus of the present study is on the fundamentals of resource use and environmental impact in the sheep industry, specifically the utilisation efficiency of land, water and energy use, and assessment of the environmental impacts 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, in the long term environmental sustainability will be constrained by the availability of these resources, and utilisation efficiency is a key measure of 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 contribute to a global phenomenon with numerous causes and uncertain effects.

Additionally, there is a very weak link between cause and effect at the local level, making it hard to 'see' the impact of emissions from a given farm. None-the-less, such assessments must be made, because agriculture can have a significant contribution to overall impacts when whole industries (rather than individual farms) are taken into account.

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 lamb 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 (e.g. tractors, harvesters), 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 none-the-less 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 (2013b, 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 (ABS 2013b). 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 (e.g. 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 'consumptive water use' can broadly be related 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 in one of the more comprehensive water footprint studies for global livestock (Mekonnen & Hoekstra 2011, 2012).

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 if there is 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, 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 described this method and apply the term 'stress-weighted water footprint', with units of L H_2O-e. The stress weighted water use applies different stress weighting factors for different regions of Australia. To calculate the stress weighted water use, consumptive water use in each region multiplied by the relevant WSI and summed across the supply chain. The value is then divided by the global average WSI (0.602) and expressed as water equivalents (Ridoutt & Pfister 2010). Using this approach, Ridoutt et al. (2012) estimated that the stress weighted water use for lamb cuts produced from south-west Victoria, transported and consumed in the western USA was 44 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 cultivation (arable). With a growing demand for food and biofuel production from the world's land resources, utilisation efficiency is an increasingly important factor, though there is a general lack of consensus on how this should be measured in LCA. To date, most LCAs have reported simply the total land required over a given time period (e.g. m^2 yr) by a production system with no description of the type of land used, or the impact of using that land. However, land types differ in productivity and suitability for cultivation and this needs to be taken into account to provide meaningful results. The extensive review of beef, pork, chicken, egg and milk LCA studies by de Vries & de Boer (2010) showed that beef production (the only ruminant included in the analysis) required 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 in terms of land occupation. Ruminants (cattle and sheep) can graze grass from non-arable land, while other species require grain produced on arable land. They also note that poultry and pigs require grain which could be fed directly to humans, while red meat production may not. Clearly, total 'land use' is not very informative when discussing the efficiency of food production for ruminants; greater detail is required.

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

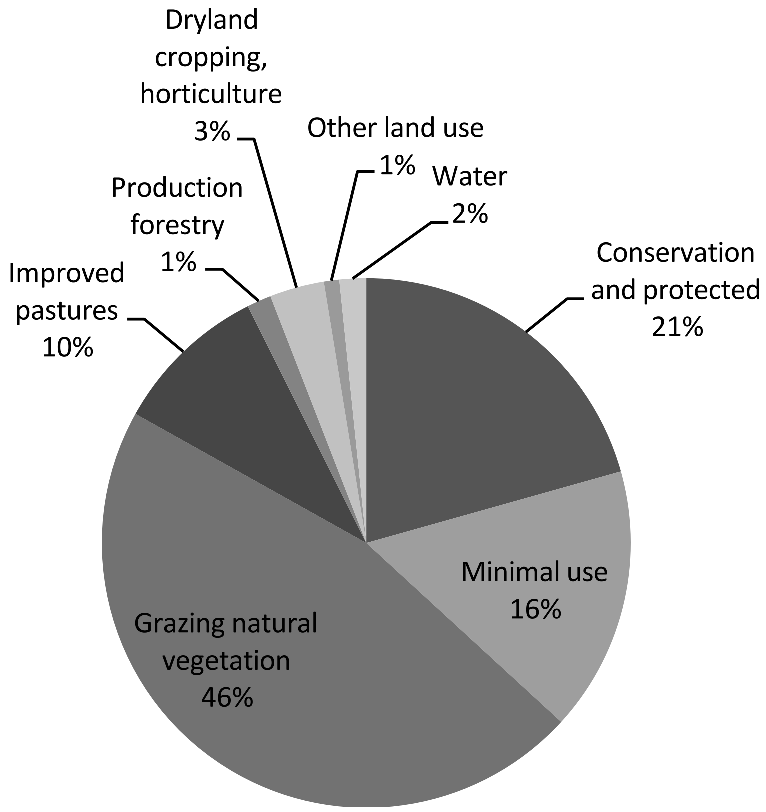


Figure 2 – Major land uses in Australia based on the 2005-06 dataset (BRS 2010)

These pastures utilised for grazing, are primarily located in the pastoral zone which is generally unsuitable for other forms of agricultural production due to soil type, topography and climate limitations. In addition, most is protected by legislation to restrict land use change. In Australia, arable land used for cropping represents only 3.4% (0.26 M ha.) of total land mass (BRS 2010). Consequently, this is a much more limiting resource and is subject to a much higher degree of competition for food production uses. 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.

1.3.4 Land use change

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 effective rainfall. Land use mapping by the Australian Bureau of Rural Sciences (BRS 2010) shows that in the five year period from 1996/97 to 2001/02, the area of land with natural vegetation used for agricultural production fell by 12.7 million ha. 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 (ISO 2013, PAS 2011). 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 conversion efficiency (HEP-CE)

Grain is an important primary commodity which can be used directly for human consumption or contribute indirectly to human food supply 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, with differences between the species arising from fundamental physiological differences. In particular, monogastrics (poultry and pigs) have a 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 dry matter / weaned pig, and produce 20-24 sale pigs per sow per year (Wiedemann et al. 2012). In contrast, a ewe may consume 500-600 kg of dry mater per lamb weaned (not accounting for the feed consumed by the lamb). It is also typical for breeding ewes to produce fewer than two lambs per ewe on average across a flock. However, one very important difference exists. Ruminants consume grass, which has a very low level of digestibility for monogastric animals and is not suitable for human consumption. Consequently, the FCR is not directly comparable without taking into consideration the ‘human edible’ portion of FCR. CAST (1999) reported the ratio of human edible protein and energy output from livestock products to human edible protein and energy input consumed by livestock as a way of quantifying the contribution or conflict between animal production and food supply. The ‘human edible protein conversion efficiency’, or HEP-CE of a livestock system, is informative to the discussion of animal agriculture’s contribution to food supply. HEP-CE is the inverse of human edible protein FCR. Wilkinson (2011) modelled the human edible protein FCR for UK livestock systems based on CAST (1999). The inverse of human edible protein FCR gives the HEP-CE (Table 1), the metric reported here. The HEP-CE values can be understood as follows: values higher than one (1) indicate more human edible protein is produced than consumed. Values lower than one indicate that more human edible protein must be consumed by the production system than what is produced. Values >1 indicate increasing efficiency of protein production because the system relies on a high proportion of protein requirements from grass. Because some systems use only small amounts of grain, these values can be much greater than one.

Table 1 – Comparative human edible protein efficiency of livestock production systems in UK (modified from Wilkinson 2011)

	Total protein efficiency (kg output from animal product/kg input of feed)	Edible protein efficiency(kg output from animal product/kg input of feed)
Raw milk	0.18	1.41
Upland suckler beef (18 mths)	0.04	1.09
Lowland suckler beef (20 mths)	0.04	0.50
Dairy bred beef (18-20 mths)	0.07	0.63
Upland lamb	0.03	0.63
Lowland lamb	0.03	0.91
Pig meat	0.23	0.38
Poultry meat	0.33	0.48
Eggs	0.31	0.43

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. Lamb 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 (e.g. grain, animal health products). 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 assessment methods.

2 Methodology

2.1 Goal definition

This project aimed to provide robust analysis of the sustainability of Australian lamb exported to the USA. This product is predominantly sourced from the south-eastern states in Australia. The study aimed to provide robust data to inform the general public, the sheep industry and the science community.

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 retail ready (bone-in) Australian lamb 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 lamb through to the point of cold storage on the east coast of the USA (Figure 3). At this point, impacts through to consumption will not be greatly different regardless of the origin of the lamb.

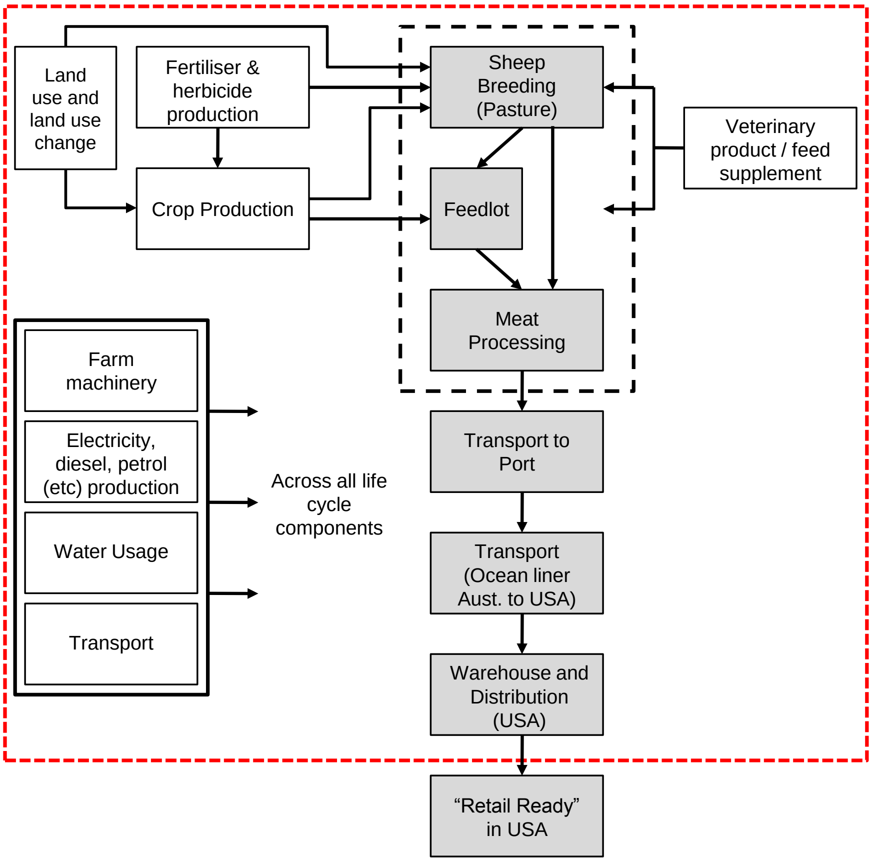


Figure 3 – Generalised system boundary for one kilogram of Australian lamb produced and exported to the USA

2.3 Impact categories and methods

The study included assessment of six 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

Energy demand was assessed using the fossil fuel energy demand (Frischknecht et al. 2007), 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). Descriptions of these methods are provided in Table 2, and detailed inventory methods are provided in Appendix 2.

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. Water retained in product was modelled as a consumptive use, assuming that the final flow following product consumption would be an evaporative loss.	Degradative water uses were assessed to be relatively minor for the production systems of interest and were not included.
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 (Pfister et al. 2009)	

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 land occupation from all stages in the supply chain and from background processes. The farm stage dominated land occupation and to improve the resolution of these results, we reported 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. Land occupation associated with other (industrial) uses was included as a residual.

At each farm, the proportion of land in each category was determined from information provided by the farmers, field observations and from analysis of aerial photography or satellite imagery where available. 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. Fallow time was assigned to the main crop. We assumed no double-cropping in the CSF or RAF, which is a reasonable assumption as there is very little double cropping in dryland regions of Australia.

2.3.4 Human edible protein conversion efficiency (HEP-CE)

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 efficiency was determined by dividing the protein content (0.19) of lamb by edible protein in the feed consumed. Additionally we took into account the yield of co-products in the avoided product system that substitute for human edible protein sources indirectly via their interaction in the animal feed and pet food markets.

2.3.5 Greenhouse gas emissions

Greenhouse gas emissions were determined from all sources relevant to lamb 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). Emissions related to fossil fuel energy use were determined from the inventory of purchased inputs and direct fuel use. Emissions of refrigerants were included for the shipping component but were not available for meat processing because of a lack of data. Emissions from Land Use and direct Land Use Change (LUdLUC) were included where relevant within the on-farm component of the supply chain. Livestock and LUdLUC emission prediction methods are outlined in Appendix 3.

2.4 Life cycle inventory

2.4.1 Supply chain characteristics

The majority of prime lambs produced for the USA market are drawn from the southern and eastern states. This study was based on three major production regions; Victoria (VIC), south-east South Australia (SA) and northern and southern New South Wales (NSW). Collectively VIC represents ~21% of Australia's sheep flock, south-eastern SA represents ~15%, while northern and southern regions in NSW represent a further ~37% of the Australian flock (MLA 2012). Australian sheep meat exported for the USA market are primarily produced from pastures, though a small segment of the market is grain finished. The present study investigated pasture fed (grass fed) lamb production from each of the three regions, and additionally included a system where lambs were finished on grain for 46 days in Victoria and South Australia. Grain finished lambs were modelled to represent 15% of production in Victoria and South Australia.

2.4.2 Farm inventory data

Foreground data were accessed for case study farms (CSF) from farm financial accounts and production records. Assessments were also made of farm biophysical resources. Regionalised data were sought from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) from surveys of specialist lamb producers in each region (regional average farms, or RAF). Where this dataset did not provide adequate detail (such as regarding water supply sources) data were substituted from the case study farm dataset.

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, flock characteristics and livestock performance. Similarly, important outputs such as enteric methane emissions could not be measured, but were modelled based on flock data.

The production supply chains were modelled using data from a broad industry survey of 203 specialist lamb producers (regional average farms – RAF) and a detailed, farm analysis of 10 case study farms (CSF) to improve the analysis of specific aspects such as water use. Farms and regions were selected to provide coverage of key production zones, with production systems suitable for supplying the USA market. Industry survey data were sourced from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) survey of specialist lamb producers, which are defined as those producers who sell more than 200 lambs annually and derive >20% of income from lamb sales. These data were averaged over the five year period 2006-2010 (ABARES 2013). From these data, a flock model was constructed and inputs associated with lamb production were determined. CSF inventory data were collected for a 1-2 year period and key productivity data were averaged over a longer period (minimum 2 years) to improve the representativeness of the dataset.

Lamb growth rates were not available from the industry wide survey. However, lamb sale values were available, and these were used to estimate final weights based on average market statistics for the survey years. Sale age for lambs from the industry survey was estimated to be 9-12 months depending on region and estimated final weights. Lamb age and weight data were accurately known for the case study farms, and these provided a validation dataset for the regional survey. The water inventory was constructed from flock inventory data, farm water supply data from the case study farms, and irrigation use from the case study and regional datasets.

The regional survey did not supply specific information for grain finishing of lambs, and this information was supplied from case study lamb feedlot enterprises from Victoria and South Australia. Descriptions and abbreviations for the supply chains are provided in Table 3, and flock productivity data are provided in Table 4. Detailed inventory data are supplied in Appendix 1.

Table 3 – Description of supply chains modelled

Region	Lamb production description	Primary dataset	Abbreviation
South-west Victoria (VIC)	Pasture fed breeding and finishing in intensive rangeland areas	Data collected from five Case Study Farms (CSF) breeding second cross meat lambs	VIC CSF
Victoria	Specialist lamb producers	Production modeled from industry survey of 79 farms used to provide a Regional Average Farm	VIC RAF
South eastern South Australia (SA)	Pasture fed breeding and finishing in extensive rangeland areas	Data collected from three Case Study Farms (CSF) producing lambs from Meat Merino flocks.	SA CSF
South Australia	Specialist lamb producers	Production modeled from industry survey of 41 farms used to provide a Regional Average Farm	SA RAF
Northern and southern NSW (NSW)	Pasture fed breeding and finishing in intensive rangeland areas	Data collected from three Case Study Farms (CSF) producing first cross lambs from Merino ewes and second cross lambs.	NSW CSF
New South Wales	Specialist lamb producers	Production modeled from industry survey of 83 farms used to provide a Regional Average Farm	NSW RAF

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

Parameter	Units	VIC RAF	VIC CSF	SA RAF	SA CSF	NSW RAF	NSW CSF
Breeding ewes	No. joined	1309	2420	1171	2733	1255	2220
Breeding ewe culling rate	%	23	23	23	29	23	21
Breeding ewe mortality rate	%	3.1	4.8	3.2	4.0	4.7	3.2
Ewe standard reference weight (SRW)	kg	65	68	65	60	65	62
Lambing % (at marking)	%	99	110	101	98	92	110
Average Lamb Weight at sale	kg	53	52	48	52	54	50
ADG - lambs	kg/day	0.14	0.17	0.13	0.10	0.13	0.16
Annual lamb sales	kg LW	46629	104898	41216	66675	47144	78527
Annual sheep sales (hoggets, mutton)	kg LW	20300	31718	18129	70265	18892	40583
Annual wool sales	kg greasy	8121	17839	7780	29027	8972	13557
Live weight per breeding ewe	kg LW/hd	53	58	51	50	53	54
Biophysical allocation factor for sheep meat*	%	72	69	70	57	69	72
Economic allocation for sheep meat*	%	75	81	73	52	69	76

* Represents allocation factor for live weight and wool at the farm gate. Factor for wool is 1-sheep meat factor.

2.4.3 Meat processing, transport and storage

Primary data were collected from an industry survey of three major meat processing plants in the region of interest (GHD 2011). Transport stages were included throughout the supply chain based on representative truck types and load specifications. International transport of chilled retail ready lamb 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 meat import processes and interviews with importers. Detailed inventory data are supplied in Appendix 1.

2.4.4 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. (2010a) and Wiedemann & McGahan (2011).

2.5 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, farms may produce products other than those from the sheep flock, such as cereal grain or beef, and the impacts associated with these must be separated. Additionally, sheep produce both wool and meat jointly. Methods for handling these are described in the following sections.

2.5.1 Dividing production systems

We handled co-production of sheep products, beef and cereal grain on the farms by subdividing the farm into systems and accounting for each separately. 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 feed resources by each species relative to the total annual feed production. Overheads were apportioned to cropping based on the proportion of land cropped compared to pasture.

2.5.2 Co-production in the sheep system

Within the lamb production system, there are a range of products that are generated at different points in the supply chain. The supply chains produced meat from culled breeding animals and lambs. Only selected cuts from lambs are exported to the USA market. Much of the meat from older breeding animals, human edible offal and some lower cost cuts are exported to other markets such as the Middle East, where the market preference is for more flavoursome meat for slow cooked dishes. Impacts were divided evenly (by mass) over all human edible products from the lamb supply chain because there are no significant biophysical or nutritional differences between the products.

Handling co-production of wool and live weight is more complex, and system subdivision is not possible because wool and meat are co-produced from the sheep flock. We applied here a biophysical allocation method to derive impacts associated with meat production. This was based on an adaptation of Cronje (2012), who suggested using the proportion of Digestible Protein Leaving the Stomach (DPLS) as the biophysical basis for dividing impacts between wool and live weight. The DPLS requirements were determined using CSIRO

(2007) methods for each flock, and total requirements for maintenance, wool and live weight growth were calculated. Allocation to live weight was determined by dividing the proportion of protein used for live weight by the total utilised protein.

At the point of meat processing, impacts were also divided between meat and skins using a biophysical approach based on the mass of product. Decisions regarding co-production are described in Table 5 and specific allocation factors for meat processing are provided in Table 6.

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
Grazing farm	Lamb (cull breeders)	Product not differentiated – equivalent to mass allocation	There was no clear rationale for discriminating between sheep meat from prime lambs and cull breeders, considering the meat product from both classes of sheep 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 meat produced from all classes of saleable sheep.
Grazing farm	live weight (wool)	Biophysical allocation	Wool and meat from live weight are jointly produced from the sheep flock. A biophysical approach based on protein requirements for wool and live weight was applied.
Meat Processing	Retail meat products (skin, edible offal)	Biophysical causality based on product mass	Allocation between meat, edible offal and skins was performed using a biophysical allocation method based on product mass. Mass was considered a reasonable proxy for the biological processes required to produce these products over the lifetime of the animal, as each is a protein based product.
Meat Processing	Retail meat products (meat/blood meal, tallow, pet food)	System expansion	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, together with economic allocation values which were applied for comparison in the sensitivity analysis. These values correspond to a dressing percentage of 45%, chilling and cutting losses of 4%, and a retail yield (bone-in retail cuts as a proportion of cold carcase weight) of 88% after the removal of fat trim. Prices used to determine the economic allocation factors are provided in Appendix 1.

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

Products	Mass of product (kg)	Biophysical causality	Economic allocation	System expansion substitution products
Retail cuts (bone-in)*	380.2	75.8%	82.6%	
Edible offal	46.4	9.3%	5.8%	
Hides	75.0	15.0%	7.2%	
Meat, blood and bone meal	69.8	n/a	1.6%	Australian ‘market average’ soymeal based on domestic production (10%) and imported product from the USA (40%) and Brazil and Argentina (50%).
Tallow	81.1	n/a	2.3%	Palm oil – Malaysia
Pet food	15.0	n/a	0.5%	Australian ‘market average’ soymeal based on domestic production (10%) and imported product from the USA (40%) and Brazil and Argentina (50%).
Totals	667.5	100.0%	100.0%	

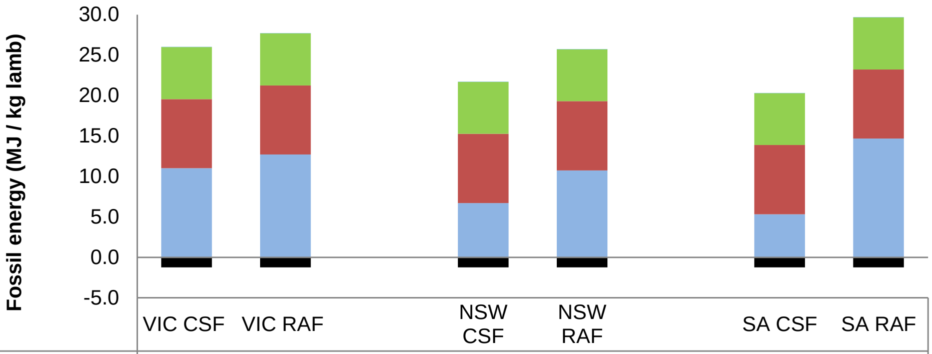
* Edible yield of retail cuts (meat less bone) is 0.8 kg / kg.

3 Results

3.1 Resource use

3.1.1 Energy demand

Total fossil fuel energy demand ranged from 19.1 to 28.5 MJ per kg lamb (Figure 4, note the negative flows reduce the total values apparent in the figure). The largest contribution was from the farm production stage (averaging 43%), followed by meat processing (35%) and transportation (26%). A small negative flow of energy use (averaging -1.2 MJ per kg lamb) relates to the substitution process applied for handling meat processing co-products. 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.

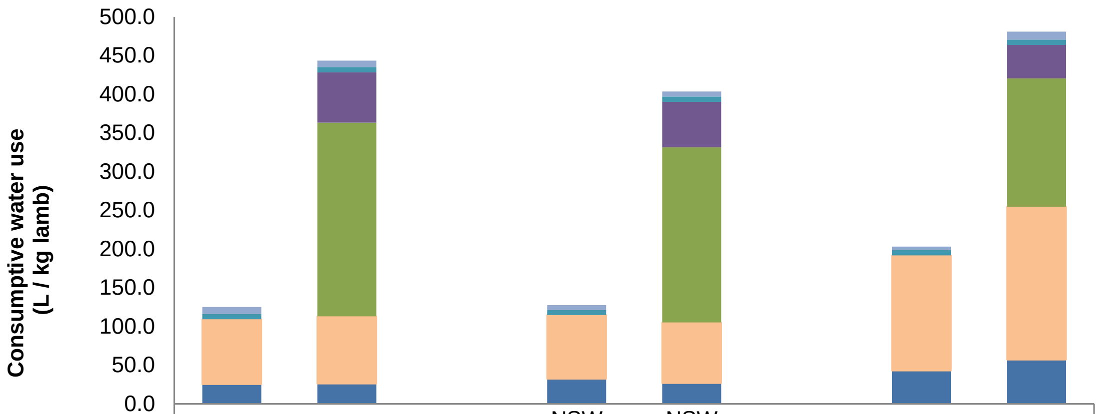


	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Warehouse	0.03	0.03	0.03	0.03	0.03	0.03
Transport	6.4	6.4	6.4	6.4	6.4	6.4
Avoided production	-1.2	-1.2	-1.2	-1.2	-1.2	-1.2
Meat processing	8.6	8.6	8.6	8.6	8.6	8.6
Farm	11.0	12.7	6.7	10.7	5.3	14.7

Figure 4 – Contribution of processes to fossil energy per kg of lamb from VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

3.1.2 Consumptive fresh water use

Consumptive water use ranged from 125.2 to 481.1 L per kg lamb (Figure 5). The large variation mainly related to the variation in irrigation water between the case study farms, which did not use irrigation, and the regional average dataset which included irrigation. Irrigation (including evaporative supply losses) contributed from 0 to 315 L / kg lamb. Evaporative losses associated with the supply of drinking water on farm contributed significantly to consumptive water use (av. 113 L / kg lamb). Consumptive water use for livestock drinking, meat processing and post-processing contributed the remaining volume. The case study farm dataset, while providing a reasonable representation of flock productivity and inputs, was less representative with regards to irrigation. None of the case study farms had irrigation supplies; consequently total water use was higher from the regional average dataset which included irrigation water use across the region.



	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Other minor use	9.1	8.4	6.2	6.8	4.7	10.6
Avoided Production	0.0	0.0	0.0	0.0	0.0	0.0
Meat processing	6.8	6.8	6.8	6.8	6.8	6.8
Irrigation losses	0.0	65.0	0.0	58.8	0.0	43.1
Irrigation	0.0	250.0	0.0	226.0	0.0	165.8
Drinking losses	83.9	87.3	82.2	78.4	148.7	197.6
Drinking	25.5	26.0	32.5	26.9	43.1	57.2

Figure 5 – Contribution of processes to consumptive water use per kg of lamb from VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

3.1.3 Land occupation

Total land occupation ranged from 46.6-1363.0 m^2 / kg lamb (Table 7), with the higher value corresponding to the South Australian production systems. The much higher land occupation in the SA supply chains compared to NSW and VIC reflects much lower stocking densities in the region, which corresponds to lower rainfall and higher evapo-transpiration rates.

The disaggregated assessment of land occupation showed that total arable land (cultivated land and arable pasture combined) averaged 10.9 m^2 / kg lamb, or 3% of total land occupation.

Feed grain production was the predominant contributor to cultivated arable land occupation.

Table 7 – Land occupation per kg of lamb from VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

	Total land occupation	Cultivated arable land	Arable Pasture
		m^2/ kg lamb	
VIC CSF	46.4	2.3	11.5
NSW CSF	72.0	3.6	2.4
SA CSF	1362.8	0.8	0.0
VIC RAF	95.1	2.2	4.7
NSW RAF	418.8	1.5	20.9
SA RAF	257.9	2.5	12.8

3.1.4 human edible protein conversion efficiency

Human edible protein conversion efficiency (HEP-CE) ranged from 1.5 to 6.3, indicating that all lamb supply chains produced more human edible protein than they consumed (values >1 indicate HEP-CE production is greater than consumption). Australian lamb production is predominantly based on grass finishing with small amounts of supplementary feeding used on occasions to improve flock productivity and lamb growth rates.

3.2 Environmental impacts

3.2.1 Stress weighted water use

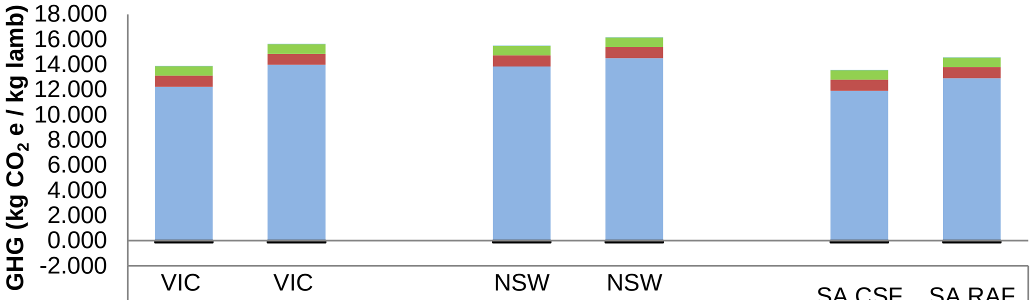
Stress weighted water use results differed considerably between regions and supply chains, ranging from 12.1 to 280.9 L H_2O e/kg lamb (Table 8). There was a wide variation in the degree of water stress in the regions modelled, from very low (0.012) to high (0.815). Weighted average values were intermediate between these levels for the regional average farms. This led to a wide range in stress weighted water use results between the regions. The stress weighted results suggested lamb produced in NSW had a greater impact on stressed water resources than lamb produced in SA, despite the latter using more water in volumetric terms.

Table 8 – Stress weighted and consumptive water use per kg of lamb from VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

Supply chain	consumptive water use	stress weighted water use
	L	L H_2O-e
VIC CSF	125.2	17.5
VIC RAF	443.5	177.5
NSW CSF	127.6	47.4
NSW RAF	403.8	280.9
SA CSF	203.3	12.1
SA RAF	481.1	23.7

3.2.2 Greenhouse gas emissions

Greenhouse gas emissions (excluding LU and dLUC) ranged from 13.4 to 16.0 kg CO_2-e per kg lamb (Figure 6). Emissions were slightly lower from the CSF dataset mainly because of the better flock productivity compared to the RAF. Contributions of components were similar between the supply chains. The GHG emissions profile was dominated by emissions at the farm, which averaged 90% of total impacts. Farm emissions were associated with enteric methane from the sheep flock which alone contributed 72-79% to total emissions. Meat processing was the second largest contributor to total GHG, with transportation of meat from Australia to the warehouse in the USA contributing only a small amount (0.8 kg CO_2-e per kg lamb) or 6% of total GHG.



	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Warehouse	0.002	0.002	0.002	0.002	0.002	0.002
Transport	0.756	0.756	0.756	0.756	0.756	0.756
Avoided production	-0.156	-0.156	-0.156	-0.156	-0.156	-0.156
Meat processing	0.884	0.884	0.884	0.884	0.884	0.884
Farm	12.234	13.977	13.853	14.512	11.911	12.920

Figure 6 – Contribution of processes to GHG emissions (excluding LU and dLUC) per kg of lamb from VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

In addition to the above GHG emissions, LU and dLUC emissions were assessed with two scenarios to reflect the higher degree of uncertainty regarding modelling assumptions. This showed LU and dLUC resulted in either a small emission source (0.3 kg CO_2 eq/kg lamb), where removals were assumed to be zero, or removals ranging from -4.0 (SA RAF) to -0.8 (SA CSF) kg CO_2 eq/kg lamb. Two main sources of removals existed. Firstly, inclusion of modest rates of soil carbon sequestration under fertilised pastures resulted in a net removal on-farm, ranging from -3.5 to 0 kg CO_2-e / kg lamb depending on pasture and fertiliser management. Additionally, a small source of avoided dLUC emissions (-0.95 kg CO_2 eq/kg lamb) also arose from the substitution of animal protein meals with soymeal imported from the USA and South America. The substitution of animal protein meal with domestic and imported soybean meal is justified in Australia where animal meals are used in the diet formulations for poultry and pigs and directly compete with imported soymeal from the USA, Brazil and Argentina.

4 Discussion

Life cycle assessment studies are reliant on modelling estimates and multiple assumptions throughout the study. To aid interpretation of the results, an uncertainty analysis and a sensitivity analysis was performed to investigate the impact of alternative assumptions throughout the model.

4.1 Uncertainty analysis

An uncertainty analysis accounting for inter-annual variation in inputs, and uncertainty related to assumptions made during the modelling process was undertaken using a Monte Carlo analysis in SimaPro 7.3. One thousand iterations provided a 95% confidence interval for results. Results and uncertainty for the three impacts most heavily influenced by modelling parameters are reported in Table 9.

Table 9 – Uncertainty and environmental impacts from lamb production

	Consumptive water use (L / kg lamb)	Fossil energy (MJ / kg lamb)	GHG emissions (kg CO_2-e / kg lamb)
VIC CSF	116.6 ± 39%	24.9 ± 19%	13.8 ± 23%
VIC RAF	435.6 ± 31%	26.5 ± 19%	15.5 ± 25%
NSW CSF	121.8 ± 35%	20.5 ± 22%	15.4 ± 27%
NSW RAF	397.3 ± 27%	24.6 ± 20%	16.0 ± 24%
SA CSF	198.9 ± 39%	19.2 ± 22%	13.4 ± 25%
SA RAF	471.1 ± 27%	28.5 ± 20%	14.5 ± 25%

Uncertainty in the consumptive water use mainly relates to uncertainty in the predicted evaporative loss rates from farm dams and irrigation. Dam evaporation rates are variable and are difficult to determine, resulting in a wide confidence interval in these results. Energy demand results were subject to a lesser degree of uncertainty, because these inputs were more readily determined from records of farm purchases rather than modelling. Uncertainty in the greenhouse gas emissions related to the underlying assumptions regarding estimation of feed intake and animal emissions. Brock et al. (2013) noted that different feed intake assumptions could result in a 20% variation in GHG emission predictions for Australian sheep, which is reflected in the confidence intervals for this study.

4.2 Sensitivity analysis

4.2.1 Handling co-production

The impact assessment results in the present study were sensitive to the methods applied for handling co-production. In the present study, different methods were applied to handle co-production at different points in the supply chain with an emphasis on the use of biophysical allocation methods at major stages and system expansion for minor co-products. This differs from previous studies, which have generally applied economic allocation despite this being the least favoured method in the ISO hierarchy. To explore the sensitivity of the allocation choices, we analysed the two most significant allocation stages (wool and live weight, and meat, hides and co-products) using economic allocation for comparison. These results also allow for a greater degree of comparability with other studies in the literature.

Compared to the preferred allocation method (biophysical allocation combined with system expansion), economic allocation resulted in a 9% increase in GHG emissions, 10% increase in fossil energy, and 8% in consumptive water. While favoured by researchers for its simplicity, economic allocation is the least favourable allocation method in the ISO standard and results are susceptible to price volatility. Additionally, economic allocation is difficult to accurately achieve in meat processing because the varied products may be at different stages in production, resulting in very different values at the point of allocation than what the finished value of the product would suggest. For example, hides are a raw product at the meat processing plant and are valued at a small fraction of the end value of the end-product leather. An alternative economic allocation approach would be to assess all products at the same point of manufacture (i.e. wholesale product) integrating additional impacts from further processing to provide a more complete analysis of the product value generated from livestock. The full results of the study using economic allocation are provided for comparison in Appendix 4.

Resource and impact analysis results are also presented for wool in Appendix 4. While not the focus of the present analysis, wool is an important co-product and allocation choices such as those made to divide between wool and live weight at the farm gate have an important bearing on the impacts attributed to both the lamb and wool product. In some cases, lamb farms produced lambs from ewes with a higher value and yield of wool from the ewe flock, resulting in different biophysical allocation results. Volatility in allocation proportions with economic allocation methods is pronounced, because wool value can vary widely between sheep breeds and from year to year.

4.2.2 GHG model assumptions

Within the GHG prediction model, enteric methane was the largest emission source for lamb production in all supply chains. This study applied Australian NGGI (DCCEE 2012a) methods for both grazing and lot-feeding stages for all supply chains. We compared the results with the IPCC Tier 2 method (Dong et al. 2006), which assumes methane yield is 6.5% ± 1.0% of gross energy intake (GEI) for mature sheep grazing pastures, and 4.5% ± 1.0% for lambs (< 1 year). This showed impacts were very similar (~4% higher) for the Australian NGGI methods. The differences associated with manure nitrous oxide between the NGGI and the IPCC default values are greater (IPCC is a factor of two higher than the Australian value) but the differences are well founded. Australian conditions do not favour nitrous oxide emissions; rainfall tends to be lower (500-750 mm in the main lamb production regions) than many other parts of the world, and pan evaporation is very high, exceeding 2000 mm in some regions (e.g. SA). Consequently, soil conditions are dry and nitrous oxide emissions are lower than may be expected in wetter climates. This is reflected in the lower Australian nitrous oxide emission factors for manure (0.004 and 0.005 kg N_2O-N/kg N for urinary and faecal N respectively) compared to IPCC defaults (0.01 kg N_2O-N/kg N).

Emissions or removals from soil carbon are less well understood than other emission sources for sheep production, with no published sheep LCAs addressing these losses to the author's knowledge. Higher dLUC emissions may exist from crop land because of variation in soil carbon loss, though this will only have a small impact on lamb because of the small amount of cultivated land required for lamb production. Alternative assumptions for soil carbon change under pasture may have a greater effect. If higher levels of soil carbon sequestration such as the 9.9 t C/ha reported by Chan et al. (2010) could be achieved, removals could be ~30% higher than reported in the 'high' sequestration scenario. However, this is unlikely to be observed in lower rainfall and mixed cropping regions (Davy & Koen 2013).

4.2.3 Transportation to the USA

Australian lamb 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. Importing lamb into the port of LA had a relatively small effect on GHG (-1%) and a modest impact on energy use (-7%).
A series of parameters were used to model the transportation from Australia to the USA and warehousing in USA, including distance from farm to port, storage time at warehouse, and distance post-warehouse. We applied a conservative shipping unit process, after Webb et al. (2013) and assumed that ships carried goods both ways, because Australia operates a large trade deficit with the USA. A sensitivity analysis was performed to evaluate the baseline assumptions. These were; farm to port distance was 350 km, warehouse storage was 80 L*yr, and post warehouse distance was 600 km (50% of supply being transported 200 km and 50% being transported 1000 km). Sensitivity results are presented as difference from baseline for transporting one kilogram of product (Figure 7). Storage time at warehouse has little effect on the GHG and energy use due to the small contribution of warehousing. Transportation distances tend to have a small effect (< 3%) on GHG, though increasing the distance of post-warehouse transport from 600 km (baseline) to 2000 km increased the overall energy use per kilogram of lamb by 8.5%. Energy consumption is

more sensitive to transport distances than GHG emissions because of the significant contribution to GHG emissions from enteric methane.

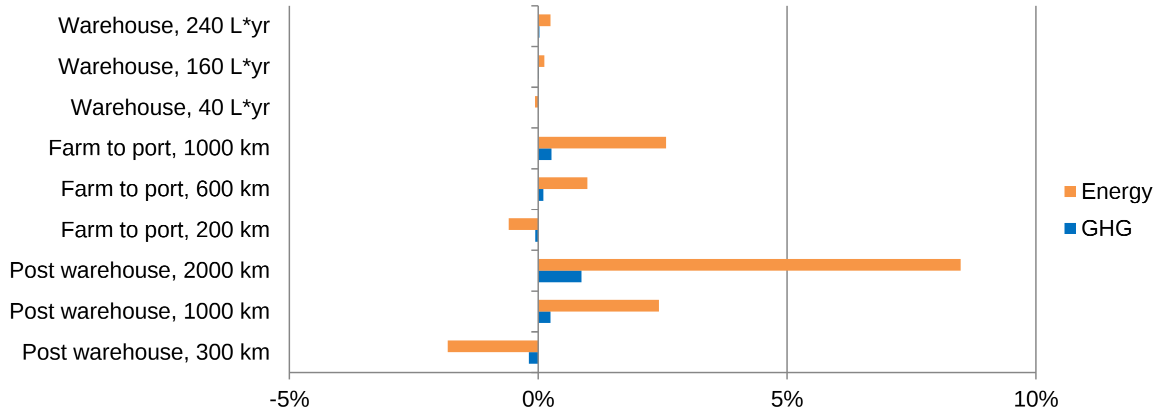


Figure 7 – Sensitivity of GHG and energy towards transportation and warehousing of one tonne of product. Results are presented as difference from baseline

4.3 Comparison with the literature

This study and others found impacts from lamb production to be highest for the 'cradle to farm gate' stage of production for GHG (Ledgard et al. 2011, Peters et al. 2010a) and water (Ridoutt et al. 2012). Consequently, this was considered an appropriate point to explore variability and points of consensus between studies. Results from the present study, presented per kilogram of live weight at the farm gate, are reported in Table 10.

Table 10 – Impacts per kilogram of live weight produced at the farm gate for grass fed lamb produced from VIC, NSW and SA. CSF = case study farms, RAF = regional average farms

	per kg LW	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Global Warming	kg CO_2-e	6.1	7.0	7.0	7.3	6.0	6.5
Fossil energy	MJ	5.5	6.4	3.4	5.4	2.7	7.4
Consumptive Water Use	L	55.0	215.1	57.7	195.9	96.3	232.9
Stress weighted water use	L H_2O-e	5.6	121.8	20.7	137.8	2.9	62.8
Cultivated arable land	m^2	1.4	1.3	2.0	1.0	0.6	1.5
Total Land Occupation	m^2	23.6	48.1	36.5	210.4	684.1	129.7

*n.r = no data available

4.3.1 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 LCA studies have been completed for Australian sheep, covering GHG emissions (Brock et al. 2013, Eady et al. 2011), water (Ridoutt et al. 2012) or multiple impacts (Peters et al. 2010a, Peters et al. 2010b). These studies used different specified end points, and all focussed on case studies only. Differences in methodological choices at the inventory level make comparison between LCA studies difficult. Notably, each of the other Australian studies investigating sheep production applied economic allocation for dividing impacts between wool and live weight. Two studies (Brock et al. 2013, Eady et al. 2011) emphasised wool production more heavily and results from these studies are less informative to the present analysis of prime lambs. Similarly, Peters et al. (2010a) studied trade lambs produced in Western Australia, which is a different market than the present study. Only Ridoutt et al. (2012) studied a similar production system and market type to the present study, though these authors only presented results for water use.

Using economic allocation (32% to meat), Brock et al. (2013) found that the GHG emissions were 5.3 kg CO_2 eq / kg LW for wethers and cull ewes from a Southern NSW wool-sheep flock. Using economic allocation (10.8% to wethers), Eady et al. (2011) found 6.2 kg CO_2 eq / kg LW for wethers from a mixed farming system (crop, wool, stud rams) in Western Australia. Eady et al. (2011) applied an alternative biophysical allocation (quite different to the one applied in this study) and showed impacts to meat were even lower, at 3.7 kg CO_2 eq / kg LW (Eady et al. 2011). Significant differences have been found between the two studies and the current study in terms of production systems (wool vs meat sheep), GHG models and particularly, allocation procedure. For example, the stud ram production in Eady et al. (2011) took a large share of the burden due to the high economic contribution, which reduced the impacts allocated to meat products, as did the high value of wool. These results were slightly lower than the mean (6.6 kg CO_2-e / kg LW) of our results, though both were within the confidence interval of 4.9 to 8.2 kg CO_2-e / kg LW.

Ridoutt et al. (2012) investigated the consumptive and stress weighted water use from lamb products produced from three farms and two feedlots in south-west Victoria and transported and consumed in US (cradle to retail). Ridoutt et al. (2012) reported farm gate consumptive water use of 149 L / kg lamb, which was within the range of results reported here but considerably lower than the average of the RAF results. The main two differences were as follows; for the case study farms, our water balance modelling showed considerably higher volumes of water being intercepted from farm dams because of evaporation and livestock drinking. Ridoutt et al. (2012) did not appear to apply a detailed dam water balance model based on actual farms to predict evaporation, which may have resulted in a lower level of predicted evaporation to the present study. We found the ratio of evaporation to water utilised to vary widely between farms (data not shown). Additionally, we modelled all drinking water as a removal from the hydrological system regardless of whether this water was ‘returned’ to pasture in manure or urine, as modelled by Ridoutt et al. (2012). We applied this assumption on the grounds that water returned to pasture in the form of urine or manure effectively acts as irrigation, as the water was withheld from runoff in a rainfall event, and returned to the pasture sometime after where the majority evaporates soon after excretion from the animal.

In contrast to the case study farms reported by Ridoutt et al. (2012) and in the present study, we showed large volumes of water may be used for irrigation. Because large volumes of irrigation water may be used by only a small subset of the industry, a large dataset (as provided by the ABARES) and comparison to catchment water balance modelling (ABS 2012a) the preferred method applied in the present study to estimate the expected water use from lamb production in Australia’s major regions. Ridoutt et al. (2012) reported stress weighted water use of 44 L H_2O-e / kg lamb for a whole supply chain through to consumption in the USA. In the present study, we found stress weighted water use to range between 12.1 to 280.9 L H_2O-e, depending on consumptive water use and the level of water stress in the regions of interest. The higher levels can be explained by the inclusion of irrigation water in the regional farm analysis (not present in the case study farms studied by Ridoutt et al. (2012) and by the production of sheep from water stressed regions in NSW and Victoria. The variability was similar to that found by Ridoutt et al. (2011) for beef production in NSW, which varied from 3.3 to 221 L H_2O-e / kg LW beef.

4.3.2 International studies

Comparatively fewer studies have investigated the resource use and environmental impacts of lamb compared to other meat products, with widely cited reviews such as de Vries and de Boer (2010) excluding lamb altogether. This is not surprising considering the smaller share of global meat trade from sheep compared to beef, pork or chicken, but for countries such as Australia, New Zealand (NZ) and the United Kingdom (UK), lamb production is an important industry.

Comprehensive lamb studies have been completed for NZ (Ledgard et al. 2011) and the UK (Williams et al. 2006) while a number of case studies have also been completed (i.e. Edwards-Jones et al. 2009, Ripoll-Bosch et al. 2012). More recently, the FAO (Opio et al. 2013) have also completed a global analysis of lamb meat production based on a broad modelling approach. Comparison between LCA studies is difficult because of the many methodological differences, and the following analysis is indicative only. Again, because of differences in end-point and allocation methods, we chose to explore the variability in different lamb systems primarily for the ‘cradle-to-farm-gate’ section of the supply chain where the impacts are greatest.

Ledgard et al. (2011) completed a detailed, comprehensive analysis of the carbon footprint (single impact only) of NZ lamb. This study covered all regions in the country and over 400 farms, which was more comprehensive than the 203 farms covered by the regional average analysis in the present study. As with the present study, Ledgard et al. (2011) applied detailed methods for predicting feed intake and livestock GHG emissions, which dominated the contribution analysis of their study. Ledgard et al. reported a national average of 8.6 kg CO_2-e / kg lamb LW at the farm gate. Perhaps the largest methodological difference was the application of economic allocation for handling the co-production of wool, lamb and meat from cull breeder ewes in the NZ study, in contrast to biophysical allocation (wool and live weight) and no differentiation between lamb and meat from cull breeding animals in the present study. In the present study, the difference between using biophysical allocation and economic allocation for wool and live weight resulted in 5% higher impacts for economic allocation at the farm gate and this would be greater if economic allocation was also applied for cull breeding animals. Interestingly, these results are of a similar order despite the different climate, breeds and production systems between the countries. Ledgard et al. (2011) extended their analysis to the full supply chain for lamb exported to the UK which was similar to the present analysis in some respects. Ledgard et al. (2011) included retail and cooking impacts while the present study ended at the warehouse. Ledgard et al. (2011) applied economic allocation for handling by-products from meat processing. Despite these differences, the value of 19 kg CO_2-e / kg lamb reported by Ledgard et al. (2011) is of a similar order to the 16.0 kg CO_2-e / kg lamb exported to the USA (economic allocation) in the present study. Zonderland-Thomassen et al. (2013) reported stress weighted water use from the same sheep dataset as reported by Ledgard et al. (2011) and found water use to be very low 0.1 L H_2O-e / kg LW. This study did not report consumptive water use and it was not clear what was included in the inventory. However, because of the very low WSI values for NZ, it is not surprising that stress weighted water use was low from this study.

Williams et al. (2006) conducted an analysis of multiple crops and livestock species produced in the UK using a modelling approach based on production representative of different regions in the UK. Results were reported for non-organic and organic lambs for GHG emissions and primary energy use. When presented on a LW basis using the dressing percentage reported by the authors (47%) and using adjusted GWP values, the impacts for non-organic lamb was 7 kg CO_2-e / kg LW. Primary energy use was 12 MJ / kg LW which was higher than the average of the present study. Williams et al. (2008) did a comparative analysis of NZ lamb imported to the UK and lamb produced in the UK to determine the impacts of transport. This study identified impacts of 14.1 and 11.6 kg CO_2-e / kg lamb for

UK and NZ lamb respectively. Again, these authors applied economic allocation and reported lower loss factors for lamb processing than in the present study.

Opio et al. (2013) completed a global analysis of ruminant systems, using LCA to report the GHG impacts from production of beef, lamb and goat meat. This study was based on a broad analysis of world regions and could not be compared readily with national studies or case studies. The reported impacts for “Oceania”, which includes NZ and Australia, was 15 kg CO_2-e / kg carcase weight (CW). This was higher than the average (13.9 kg CO_2-e / kg CW, using the same dressing percentage) for the present study without transport or warehousing impacts, reported using economic allocation. The contribution analysis from Opio et al. (2013) suggests much higher contributions from nitrous oxide than the present study. Considering the lower emission factors for nitrous oxide in Australian conditions (DCCEE 2012a) these results may over predict total impacts from Australian production by 10-12%.

Ripoll-Bosch et al. (2012) compared GHG emissions of lamb production from pasture based, sheep-cereal, and zero grazing systems in north-eastern Spain. They applied economic allocation between live weight of lambs sold (22 kg/head), and the cultural-ecosystem service provided from the lamb production systems based on European agri-environmental subsidies. Wool and meat from culled ewes and rams were excluded due to small economic output. Without allocation, the GHG intensity was 25.9 kg CO_2-e per kg live weight of the pasture system. When applying economic allocation between live weight output and cultural-ecosystem service (57%), the GHG intensity was 13.9 CO_2-e per kg live weight. The very different approach to allocation in this study is interesting, but quite atypical. Failing to allocate any impact to cull breeding animals was also surprising, particularly if these animals are sold for human consumption. The GHG impacts were very high in comparison to other authors or the present study, and this will largely be in response to differences in allocation and the production system. The contribution of CO_2, CH_4 and N_2O from Ripoll-Bosch et al. (2012) was 8%, 62% and 31% for pasture based lamb production, which showed a much larger contribution from nitrous oxide than the present study. These may be explained by the lower nitrous oxide emissions from manure and feed production in Australia which is partly the result of less favourable conditions for nitrous oxide emissions in Australia.

Edwards-Jones et al. (2009) analysed the carbon footprint of lamb production from two Welsh farms, reporting a wide range of emissions from 8.1 to 143.5 kg CO_2-e / kg live weight. The extremely high numbers reported were caused by nitrous oxide emissions from peat soils, which represents a unique situation not experienced in Australia.

While not directly comparable, water use results have also been presented in the water footprinting literature for lamb production in different parts of the world. Many methodological differences exist making comparisons impossible. This said, they are interesting from the perspective of considering other information on this topic which is available in the literature. Mekonnen and Hoekstra (2012) reported blue water use of 312-593 L / kg sheep meat for various countries. This value did not include supply losses, which we found to be considerable. However, these results were of a similar order to those presented for the regional assessment in this study. For interest, the total virtual water content (VWC) for sheep meat reported by Mekonnen & Hoekstra (2012) was 2839-12,240 L / kg sheep meat. VWC results are of limited value for assessing the impact of water use on competitive users or the environment. However, VWC values for red meat have been widely disseminated and the differences in analysis method must be understood. The large difference between VWC results and consumptive water from an LCA analysis relates to the inclusion or exclusion of rainfall that contributes directly to pasture or crop growth for livestock production. This soil stored moisture from rainfall, or ‘green’ water is spatially constrained and restricted to extraction by plants only. Thus, it cannot directly contribute to water supplies for industry, human use or most other agricultural uses if constrained by land type. In contrast, consumptive and stress weighted water use provide a clear analysis of water that can be utilised for competitive purposes or the environment. Stress weighted water use goes further, by providing an indication of the impact of using this water compared to the global 'average' water stress levels for global water reserves. This type of analysis is more directly useful for determining the impact of using water.

5 Conclusions

Australia is the second largest global exporter of sheep meat in the world after New Zealand. The Australian sheep industry maintains a strong emphasis on producing lamb from sustainable production systems, predominantly from the rangeland areas of south eastern Australia. This study was the first of its type to present a broad suite of resource and environmental impact indicators for Australian lamb production, processing and transport to the USA. Results from the present study were of a similar order to previous Australian LCA results when compared at the farm gate stage. Results per kilogram of lamb were difficult to compare with the literature because of differences with the level of accounting for trimming during processing and differences in allocation methods. When these differences were minimised to the greatest extent possible, greenhouse gas emissions were slightly higher than those previously reported in the Australian literature, though most previous studies investigated predominantly Merino sheep systems with high value wool which lessens the impacts contributed to meat. Full supply chain results were of a similar order to other studies. Water results using LCA have been reported by two other authors for an Australian supply chain, and the results presented here suggest that consumptive and stress weighted water are considerably higher when irrigation is taken into account. Consumptive and stress weighted water use was much lower when assessed using the LCA approach compared to virtual water. Few energy use or land occupation results have been reported previously for Australian lamb.

Lamb is a globally traded product, and concerns may exist regarding the impacts of transport on the environmental impact of lamb. The study confirmed the findings of Ledgard et al. (2011) and Williams et al. (2008) by showing that GHG impacts from long distance transport are relatively minor (≤6%). The contribution of transport to energy use was higher (averaging 27%) but the contribution of transport to other impacts was negligible.

This research has extended the analysis of case study data to include regions (the states of NSW, VIC and SA) which produce most of the lamb exported to the USA. While these broad datasets preclude some detail regarding market type, they are much more extensive and do provide a robust comparison of variable impacts such as irrigation water use. This provides confidence in the representativeness of this analysis across most impact categories. Considering the greater specificity in the case study datasets, these may be more representative of lamb exports to the USA for the impact categories GHG and energy use, while the regional dataset is more representative for water use.

This study presents a robust analysis of the lamb industry, the first multi-impact study of its kind. The study has applied new methods for handling the co-production of wool and live weight, and co-products from meat processing. These methods are considered more robust and are more closely aligned with the ISO standards for LCA.

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Appendix 1

Uncertainty

All inventory data are reported with an indication of uncertainty. Uncertainty was determined using the pedigree matrix system (Weidema & Wesnæs 1996), which was used for most inputs from the technosphere (i.e. electricity, fuel) and water inputs.

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 commodity transport to the farms. 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 as per tonne of DMI consumed (Table 11, Table 12). 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.

Table 11 – Material inputs and outputs for VIC, SA and NSW case study farms (CSF)

			VIC CSF		SA CSF		NSW CSF	
Inputs	**Data source**	**Units**	**per tonne DMI**	**Uncertainty (SD)**	**per tonne DMI**	**Uncertainty (SD)**	**per tonne DMI**	**Uncertainty (SD)**
Feed	Farm records							
Lupins		kg	0.0	1.1	0.0	1.1	8.0	1.0
Hay		kg	12.8	1.1	0.0	1.0	0.0	0.4
Cereal grain		kg	6.1	1.1	0.0	1.1	36.3	1.1
Energy	Farm records							
Electricity		kWh	2.2	1.1	3.3	1.1	2.6	1.0
Oil		L	0.03	1.1	0.0	0.0	0.02	1.4
Diesel		L	4.6	1.1	2.4	1.0	1.2	1.0
Petrol		L	0.3	1.1	0.7	1.0	0.9	1.0
Fertilisers	Farm records							
Superphosphate		kg	19.1	1.1	0.0	1.1	14.2	1.0
Pesticides	Farm records	g	146.8	1.1	0.0	0.0	81.6	1.1
Other inputs and services	Farm records							
Veterinary services		$	9.8	1.1	2.7	4.4	6.2	3.6
Communication services		$	0.8	1.3	0.5	4.4	0.5	3.6
Insurance		$	2.1	1.3	1.6	4.4	1.3	3.6
Accounting		$	1.2	1.3	0.7	4.4	0.6	3.6

Table 12 – Material inputs and outputs for VIC, SA and NSW regional average farms (RAF)

			VIC RAF	**SA RAF**	**NSW RAF**	**Uncertainty**
Inputs	**Data source description**	**Units**	**per tonne DMI**	**per tonne DMI**	**per tonne DMI**	**(SD)**
Feed	ABARES					
Lupins		kg	4.2	5.5	5.2	1.2
Hay		kg	8.1	10.5	10.0	1.2
Cereal grain		kg	4.2	5.5	5.2	1.2
Energy	ABARES					
Electricity		kWh	1.7	2.4	2.2	1.2
Oil		L	0.2	0.2	0.2	1.2
Diesel		L	5.2	7.0	4.6	1.2
Petrol		L	0.8	1.0	0.8	1.2
Fertilisers	ABARES					
Superphosphate		kg	27.5	28.4	15.0	1.2
Pesticides	ABARES	g	325.3	893.3	231.4	1.2
Other inputs and services	ABARES					
Veterinary services		$	5.85	4.2	4.8	5.2
Communication services		$	0.84	0.5	0.7	0.7
Insurance		$	1.4	2.0	1.9	1.2
Accounting		$	0.8	1.1	1.0	1.2

Feedlot inventory data

Feedlot inventory data were collected from one lamb feedlot located at south west Victoria. Additional comparison data were also collected from a South Australian lamb feedlot (data not presented here). GHG emissions were estimated from feed and lamb performance data following Australian NGGI (DCCEE 2012a), 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 except veterinary expenses. Material inputs and outputs for the lamb feedlot is presented in Table 13.

Table 13 – Material inputs and outputs for the lamb feedlot

Inputs	Data source	Units	Per animal finished (52 kg)	Uncertainty (SD)
Store lambs	Feedlot	kg	34.8	
Feed ration	Feedlot	kg DM	56.2	1.05
Land occupation	Feedlot			
Non arable (Feedlot)		m^2	7.5	1.24
Energy	Feedlot			
Electricity		kWh	0.23	1.05
Diesel		L	0.05	1.05
Petrol		L	0.02	1.05
Transport	Estimated transport distances for sheep and feedlot commodities	t.km	17.1	
Other inputs and services				
Veterinary services		$	1.2	1.05
Staff travel		km	0.6	1.10
Outputs				
Finished animal to abattoir		kg	52.0	

Feed milling and rations

Feed milling inventory data for the lamb feedlot were based on records kept by the feed mill on-site. These data are presented in Table 14.

Table 14 – Major inputs for feed milling

Inputs	Data source description	Units	Amounts per tonne ration
Energy			
Electricity		kWh	5.0
LPG	Data collected from feedlot	L	0.3
Diesel		L	0.7
Water	Data collected from feedlot	L	0
Transport	Est. transport distances for commodities to the feedlot	t.km	55

Feed inputs are the largest input for feedlot lamb production. Lambs 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. Commodity inputs to the rations were simplified using a substitution process (Wiedemann & McGahan 2011, Wiedemann et al. 2010b).

Data were not available for a number of minor dietary inputs. These inputs fall into two categories; products that require a low level of manufacturing and are of low cost (i.e. salt) and products that are high cost such as 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 one year. Commodity inputs for the sheep 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. Inputs averaged across the rations are show in Table 15.

Table 15 – Aggregated, simplified rations for the lamb feedlot

Commodities (protein content in brackets)	Units	Values
Barley (10%)	kg	694
Peas	kg	140
Protein Concentrate	kg	120
Lime	kg	10
Vegetable oil	kg	5
Sheep supplement (80%)	kg	30
Feed additives	kg	1.1
Total	**kg**	**1000.0**

Meat processing data

Inputs and impacts associated with meat processing were collected from three meat processing plants in an industry survey of sheep/lamb processing plants in southern Australia (GHD 2011). These data are shown in Table 16.

Table 16 – Major inputs associated with meat processing

Major Inputs	units	Per tonne carcase weight
Water use, 100% consumptive	L	5973
Energy Use		
Electricity	kWh	1185
LPG	MJ	533
Diesel	MJ	19
Petrol	MJ	14
Fuel oil	MJ	1184
Natural Gas	MJ	2346
Industrial land occupation	m^2	0.002

Total impacts from meat processing were 907.8 kg CO_2-e per tonne carcase weight processed. This value differs from the contribution analyses in the results section because they are presented with a different functional unit in the results.

Values for carcases and co-products were based on ex-processing prices in Australia as reported by the National Livestock Reporting System (MLA 2013), averaged from 2012 and 2013. These values are reported in Table 17.

Table 17 – Product and co-product values applied in the economic allocation **METHOD**

Portion	Mass (kg/1000 kg LW)	Price ($/kg)	Value ($/1000 kg LW)	Economic allocation
Retail cuts (bone-in)	380.2	6.75	2,565	82.6%
Edible offal	46.4	3.89	181	5.8%
Hides	75.0	3.00	225	7.2%
meat meal	62.2	0.68	42	1.4%
Blood meal	7.6	0.97	7	0.2%
Tallow	81.1	0.86	70	2.3%
Pet food	15.0	1.00	15	0.5%
Sum	667.5		3,105	100%

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

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

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). Ships were assumed to carry goods both ways, because Australia operates a large trade deficit with the USA (DFAT 2012).

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 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, 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 telephone and veterinary services were based on financial records from the supply chain matched with economic input-output tables. Impacts associated with services are typically very small; however this approach provided a comprehensive coverage of these impacts and was therefore included for completeness.

Appendix 2

Water 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 lamb.

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)

Background system for farms:

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

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% such as fertiliser and electricity production) 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 sheep, 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 by Luke, cited in CSIRO (2007), were applied in the present study. Table 19 provides climate data relevant to the farm water modeling.

Table 18 – Summary of site data used in water modelling for the case study farms and feedlot

Region	Regional average Rainfall * (mm / yr)	Regional average Evaporation * (mm / yr)	WSI *
VIC CSF	621	1168	0.0107
VIC Lamb Feedlot CSF	426	1752	0.021
SA CSF	308	2409	0.017
NSW CSF	663	1570	0.28

* Weighted average rainfall, evaporation and WSI values reported where relevant.

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 19).

Table 19 – Sources of water supply for case study farms (CSF)

Source of water supply	VIC CSF	SA CSF	NSW CSF
		% of total water supply	
Dam	72	27	56
Creek/River	9	0	17
Bore	19	73	28

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 were modelled using farm dam water balances constructed from long term 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 from the dam surface was estimated by applying a factor to the Class A pan evaporation, varying from 0.75-0.9, which were similar to values suggested by Burman & Pochop (1994) and Craig (2006).

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. The intercept to extraction ratio is based on the volume of water intercepted from the environment as a result of dam construction, which is the difference between catchment runoff volume and overflow volume. Higher ratio values indicate a greater volume of water intercepted from the environment to provide water for livestock. This ratio was strongly influenced primarily by annual evaporation rates and dam demand factors (the proportion of dam volume extracted annually). Dam characteristics, water balances and intercept ratios are shown in Table 20.

Table 20 – Farm dam characteristics for the three regions

Dam characteristics	VIC	SA	NSW
Dam volume for sheep water supply – ML	129	26	77
Dam density - ML per km^2	7.3	0.3	6.4
Demand factor	0.09	0.10	0.07
Intercept to extraction ratio	5.7	13.3	5.6

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.

Irrigation

Irrigation water was included in the inventory from two sources; direct irrigation of pastures for sheep grazing, and irrigation of feed grains used by sheep production. No case study farms used irrigation. However, the irrigated land use was reported in the ABARES survey. We allocated irrigation land use from the ABARES survey following the system division process described in the allocation section (2.5.1) and applied an irrigation rate from the ABS survey of water use on Australian farms (ABS 2012b). Estimated irrigation rates were then calibrated against the ABS survey of water use on Australian farms. This survey provided three years of data (2009-2011) of irrigation water use for 'sheep and other livestock' in NSW, VIC and SA. We applied a partitioning factor of 0.9 to determine water use for sheep only, and cross checked total water use with total sheep and lamb production from the Australian agricultural commodities dataset (ABS 2013a) in order to validate our modelled results in litres per kilogram of live weight (at the farm gate). Irrigation associated with feed grain inputs was modelled using the water inventory of Ridoutt and Poulton (2009) for NSW, applied at a pro rata basis to the states based on total irrigation for cereal cropping in each state from ABS (2012b). As this reference reported irrigation water for NSW grain only, this may have introduced a small degree of error in the irrigation contribution from grain in Victoria and South Australia, though considering the relatively small overall contribution from this source the error is not large. Losses associated with irrigation water supply were determined from the ABS national water accounts (ABS 2012a), and amounted to 27.1% of total extraction from the environment for water supplied from irrigation schemes.

Water stress index values

Pfister et al. (2009) provided a global GIS dataset of WSI values with full coverage of Australia. This dataset, while coarse, provides the opportunity to use a standardised global method. The majority of catchments in Australia have very low WSI values, with the exception of parts of the Murray Darling Basin in NSW and Victoria. All farms were located and appropriate WSI values were assigned. However, for the regional datasets, which covered the whole states of NSW, VIC and SA, a different method was required. We determined the total proportion of the sheep flock in each state for the various WSI regions using the ABS agricultural commodities dataset for 2011-12, reported in Statistical Divisions (ABS 2013). From this analysis, 27% of the NSW sheep flock was located in a WSI region of 0.815 while 28% of the VIC flock was located in this zone. SA zones were not identified as water stressed, which may be an anomaly in the WSI dataset.

Appendix 3

Livestock GHG modelling

Livestock GHG modelling was done according to the Australian tier 2 method (NIR 2013), as outlined in the following sections.

Feed Intake

Potential intake is determined largely by body size and the proportion of the diet that is able to be metabolised by the animal. Potential intake (PI_{ijk} kg DM/head/day) is given by (AFRC 1990) as:

$$\boldsymbol{PI_{ijk} = (104.7 \times q_{m,ijk} + 0.307 \times W_{ijk} - 15.0) \times W_{ijk}^{0.75}/1000}$$ **EQUATION 1**

Where:

W_{ijk} = liveweight (kg)

$q_{m,\,ijk}$ = metabolizability of the diet. This is the ratio of metabolizable energy (ME) to gross energy (GE) in the diet (i.e. ME / GE). Metabolizable energy content is related to digestibility of dry matter (DMDijk), and the equation of (Minson & McDonald 1987) is used, $q_{m,\,ijk}$ = 0.00795 DMD - 0.0014 (DMD expressed as a %)

However, the actual feed intake of animals is often less than the potential intake. This can be caused by many factors, especially by low feed availability. Relative intake is defined as the proportion of potential intake that the animal will consume. The relative intake due to feed availability is given by White et al. (1983) as:

$$\boldsymbol{R_{ijk} = 1 - \exp(-2 \times (DMA_{ijk})^2}$$ **EQUATION 2**

Where:

DMA_{ijk} = dry matter availability tonnes/hectare

Note: Actual feed intake will be less than potential intake only when feed availability is less than 1.63 tonnes/hectare.

The actual intake (I_{ijk} kg DM/head/day) of a sheep is thus:

$$\boldsymbol{I_{ijk} = P_{ijk} \times R_{ijk} \times MA_{ijk=4}}$$ **EQUATION 3**

Where:

$MA_{ijk=4}$ = additional intake for milk production (expressed as a ratio, see Equation 4)

Feed intakes can increase by up to 60% during lactation (Agricultural Research Council 1980). For emissions estimates, the intake of all breeding ewes was assumed to increase by 30% during the season in which lambing occurs, based on relationships presented in (Standing Committee on Agriculture 1990).

The additional intake for milk production ($MA_{ijk=4}$) is calculated by:

$$\boldsymbol{MA_{ijk=4} = \left(LE_{ijk=4} \times FA_{ijk=4}\right) + ((1 - LE_{ijk=4}) \times 1)}$$ **EQUATION 4**

Where:

$LE_{ijk=4}$ = proportion of breeding ewes lactating, calculated as the annual lambing rates x proportion of lambs receiving milk in each season

$FA_{ijk=4}$ = feed adjustment (assumed to be 1.3)

Enteric methane

Methane production (M_{ijk} kg/head/day) is calculated using daily intake figures (I_{ijk}) via the relationship of (Howden et al. 1994):

$$\boldsymbol{M_{ijk} = I_{ijk} \times 0.0188 + 0.00158}$$ **EQUATION 5**

Where:

M_{ijk} = methane production (kg/head/day)

Manure emissions

Manure Methane

Methane production from the manure (M_{ijk} kg/head/day) of sheep is calculated as:

$$\boldsymbol{M_{ijk} = I_{ijk} \times (1 - DMD_{ijk}) \times MEF}$$ **EQUATION 6**

Where:

I_{ijk} = dry matter intake

MEF = manure emission factor (kg CH_4 / kg DM Manure) (Gonzalez-Avalos & Ruiz-Suarez 2001). The warm factor is used for QLD and NT and the temperate factor is used for all other States.

Manure Nitrous Oxide

The methodology for calculating the excretion of nitrogen from sheep makes use of the following algorithms to calculate crude protein input (CPI_{ijk}) and storage (NR_{ijk}) and from these the output of nitrogen in the faeces and urine. The crude protein intake CPI_{ijk} (kg/head/day) of sheep is calculated thus:

$$\boldsymbol{CPI_{ijk} = I_{ijk} \times CP_{ijk} + (0.045 \times MC_{ijk})} \qquad \textbf{EQUATION 7}$$

Where:

I_{ijk} = feed intake (kg DM/head/day)

CP_{ijk} = crude protein content of feed intake expressed as a fraction

MC_{ijk} = milk intake (kg/head/day) calculated as proportion of lambs receiving milk in each season x milk intake. Milk intake assumed to be 1.6 kg/day for the first three months after the birth of lambs

Nitrogen excreted in faeces (F_{ijk} kg/head/day) is calculated, using functions developed by the (Standing Committee on Agriculture 1990) and (Freer et al. 1997), as the indigestible fraction of the undegraded protein from solid feed, the microbial crude protein and milk protein plus the endogenous faecal protein, such that:

$$\boldsymbol{F_{ijkl} = \left\{0.3\left(CPI_{ijkl} \times \left(1 - \left[\frac{(DMD_{ijkl}+10)}{100}\right]\right)\right) + 0.105\left(ME_{ijkl} \times I_{ijkl} \times 0.008\right) + 0.08\left(0.032 \times MC_{ijkl}\right) + \left(0.0152 \times I_{ijkl}\right)\right\}/6.25} \qquad \textbf{EQUATION 8}$$

Where:

DMD_{ijk} = digestibility expressed as a percentage

ME_{ijk} = metabolizable energy (MJ/kg DM) calculated as: 0.1604 DMD_{ijk}-1.037 (Minson & McDonald 1987)

MC_{ijk} = milk intake (kg/day) calculated as proportion of lambs receiving milk in each season x milk intake. Milk intake assumed to be 1.6 kg/day for the first three months after the birth of lambs

1/6.25 = factor for converting crude protein into nitrogen

The amount of nitrogen retained by the body (NR_{ijk} kg/head/day) is calculated as the nitrogen retained in milk, wool and body tissue such that:

$$\boldsymbol{NR_{ijk} = (0.045 \times MP_{ijk} + 0.84 \times WP_{ijk} + (212 - 4 \times \left(\frac{EBG_{ijk} \times 100}{4 \times SRW_{ijk}^{0.75}} - 1\right) - \frac{140 - 4 \times \left(\frac{EBG_{ijk} \times 100}{4 \times SRW_{ijk}^{0.75}} - 1\right)}{1 + EXP\left(-6 \times \left(Z_{ijk} - 0.4\right)\right)}) \times \frac{EBG_{ijk}}{1000})/6.25} \qquad \textbf{EQUATION 9}$$

Where:

MP_{ijk} = milk production in (kg/day) calculated as: proportion of ewes lactating (LE_{ijk}) x milk production. Milk production is considered to be 1.6 kg/day for breeding ewes in the first three months after the birth of lambs.

WP_{ijk} = clean wool production (kg/day) based on greasy wool production per head multiplied by clean yield percentage. It is assumed that clean wool consists of 16% water and 84% protein.

EBG_{ijk} = empty body gain which is equivalent to LWG_{ijk} x 0.92

SRW_{ijk} = standard reference weight (Standing Committee on Agriculture 1990)

Z_{ijk} = relative size (liveweight / standard reference weight)

Nitrogen excreted in urine (U_{ijk} kg/head/day) is calculated by subtracting the nitrogen retained (NR_{ijk}) and the nitrogen excreted in the faeces (F_{ijk}) from the nitrogen intake such that:

$$\boldsymbol{U_{ijk} = \left(\frac{CPI_{ijk}}{6.25}\right) - NR_{ijk} - F_{ijk}}$$ **EQUATION 10**

The annual faecal (AF_{ijk} Gg) and urinary (AU_{ijk} Gg) nitrogen excreted is calculated as:

$$\boldsymbol{AF_{ijk} = (N_{ijk} \times F_{ijk} \times 91.25) \times 10^{-6}}$$ **EQUATION 11**

$$\boldsymbol{AU_{ijk} = (N_{ijk} \times U_{ijk} \times 91.25) \times 10^{-6}}$$ **EQUATION 12**

Where:

N_{ijk} = the number sheep in each State, season and class

The total emissions of nitrous oxide from the feedpad are calculated as follows:

$$\boldsymbol{Faecal_{MMS} = AF_{ijkl} \times MMS \times EF_{(MMS)} \times C_g}$$ **EQUATION 13**

$$\boldsymbol{Urine_{MMS} = AU_{ijkl} \times MMS \times EF_{(MMS)} \times C_g}$$ **EQUATION 14**

$$\boldsymbol{Total_{MMS} = Faecal_{MMS} + Urine_{MMS}}$$ **EQUATION 15**

Where:

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

$EF_{(MMS)}$ = emissions factor (0.005 and 0.004 N_2O-N kg/N excreted for manure and urine respectively.

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

Indirect N_2O emissions from Atmospheric deposition

The mass of animal waste N volatilised is calculated as:

$$\boldsymbol{M_{ijk=2} = \sum_{MMS}(AE \times MMS \times FracGASM_{MMS})} \qquad \textbf{Equation 16}$$

Where:
AE = the sum of faecal (AF_{ijk}) and urinary (AU_{ijk}) nitrogen.

MMS = the fraction of AE that is managed in the different manure management systems.

$FracGASM_{MMS}$ = the fraction of N volatilised in each manure management systems. For pasture and paddock grazing, 0.2 was used.

Annual nitrous oxide production from atmospheric deposition is calculated as:

$$\boldsymbol{E = \sum_i \sum_k \left(M_{ijk} \times EF_{ijk} \times C_g\right)} \qquad \textbf{Equation 17}$$

Where:
E = annual emissions from atmospheric deposition (Gg N_2O)

M_{ijk} = mass of animal waste N volatilised (Gg N)

EF_{jk} = 0.004 (Gg N_2O-N/Gg N)

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

Indirect N_2O emissions from Leaching and Runoff

The mass of animal waste N applied to soils that is lost through leaching and runoff is calculated as:

$$\boldsymbol{M_{ijk=2} = (MN_{soil} + UNsoil_{ik} + FNsoil_{ik}) \times FracWET_{ik} \times FracLEACH_j} \qquad \textbf{Equation 18}$$

Where:

$M_{ijk=2}$ = Mass of animal waste N lost through leaching and runoff (Gg N)

$MNsoil_{ik}$ = mass of manure N applied to soils (Gg N)

$UNsoil_{ik}$ = mass of urinary N applied to soils (Gg N), as calculated in Equation 11

$FNsoil_{ik}$ = mass of faecal N applied to soils (Gg N), as calculated in Equation 12

$FracWET_{ik}$ = fraction of N available for leaching and runoff

$FracLEACH_{j}$ = 0.3 (Gg N/Gg applied) IPCC default fraction of N lost through leaching and runoff.

Annual nitrous oxide production from leaching and runoff is calculated as:

$$\boldsymbol{E} = \sum_{\boldsymbol{i}} \sum_{\boldsymbol{k}} \left(\boldsymbol{M_{ijk}} \times \boldsymbol{EF_{ijk}} \times \boldsymbol{C_g}\right)$$ **EQUATION 19**

Where:

E = annual emissions from leaching and runoff (Gg N_2O)

M_{ijk} = mass of N lost through leaching and runoff (Gg N)

EF_{jk} = 0.0125 (Gg N_2O-N/Gg N)

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

Land use and direct land use change GHG removals and emissions

Soil carbon changes under crop and pasture soils (LU emissions and removals), were included following guidance from LEAP (2014). Soil carbon losses from cultivated land were mainly related to the use of purchased grain sourced regionally. Estimated soil carbon losses took into account the different rate of loss from conventional and zero-tillage (Chan et al. 2003, Dalal & Chan 2001), assuming multiple cultivations occur on 37% of crop land (NSW, VIC, SA – ABS 2009) with remaining land being zero-tillage. Soil carbon losses were assumed to be 0.1 t C/ha.yr for zero tillage and 0.58 t C/ha.yr for land cultivated more than once, with the latter predicted using the equation of Dalal and Chan (2001) for light textured soils (30% clay).

In contrast to cultivated soils, soil carbon sequestration rates of 0.29 ± 0.17 t C/ha.yr (15 studies) were reported for Australian pastures, where phosphorus fertiliser and lime have been applied (Sanderman et al. 2010). Chan et al. (2010) reported a carbon stock change of 9.9 t C/ha over an estimated 25-40 years for fertilised pastures in southern NSW, though such changes have not been reported for all regions (Davy & Koen 2013, Schwenke et al. 2013). Phosphorus fertiliser application was common in the sheep systems studied (Table 11) and carbon removals were explored using two scenarios: i) zero change in soil carbon under fertilised pasture, ii) or carbon sequestration of 7.2 t C/ha under fertilised pasture, based on the sequestration rate of 0.29 t C/ha.yr in Sanderman et al. (2010) over a 25 year period. The land area fertilised was determined from the total tonnes divided by a standard application rate of 125 kg/ha (NSW and SA) and 150 kg/ha (VIC) (ABS 2009) with an assumed three year fertiliser rotation. Carbon removals from sequestration were annualised over a 100 year period.

Direct LUC emissions were determined for the previous 20 years (BSI 2011, LEAP 2014) for conversion of forest to grassland or crop land and conversion of grassland to crop land. While deforestation in southern states of Australia has fallen to very low levels since 1990 (DCCEE 2012b), historic emissions must be considered. Conversion of forest to grassland attributable to sheep was assumed to be negligible, because of the dramatic decline in Australian sheep numbers from 170 M in 1990 to 68 M in 2010 (ABS 2013a), which indicate sheep production is very unlikely to be a driver of expansion of grassland in these regions. In contrast, crop land has expanded by 14%, 26% and 24% for NSW, VIC and SA respectively, based on 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 (ABS 2013a). Direct LUC was predominantly from conversion of grassland (DCCEE 2012b), assumed to be 80% (NSW) and 95% (VIC, SA) of new crop land. Total carbon losses of 84 t C/ha (forest to crop land) and 12.6 t C/ha (grassland to crop land) were assumed using tier II methods (DCCEE 2012b), corresponding to annualised emission rates of 15.5 and 2.3 t CO_2-e/ha.yr. Divided over the total cultivated land, dLUC emissions per hectare were 0.69, 0.77 and 0.7 t CO_2-e/ha.yr for NSW, VIC and SA respectively.

Appendix 4

Table 21 and Table 22 provide results using economic allocation for handling co-production at the two major points in the supply chain; live weight and wool at the farm gate, and meat, hides and co-products at the processor. Results per kg greasy wool using biophysical allocation are presented in Table 23.

Table 21 – Impact assessment per kg of live weight (LW) with economic allocation from the VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

	GHG kg CO_2-e/kg LW	Fossil energy MJ/kg LW	Consumptive fresh water use L/kg LW	Waterstress wsi eq L /kg LW	Cultivated land use m^2/ kg LW	Total land use m^2/ kg LW
VIC CSF	7.1	6.4	66.1	6.6	1.5	27.3
VIC RAF	7.3	6.7	227.3	90.0	1.4	50.3
NSW CSF	7.5	3.6	62.5	23.9	2.3	39.2
NSW RAF	7.2	5.3	194.6	136.0	1.0	207.7
SA CSF	5.8	2.6	93.0	2.8	0.6	657.9
SA RAF	6.7	7.6	243.4	9.0	1.5	133.8

Table 22 – Impact assessment per kg of lamb with economic allocation from the VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

	GHG kg CO_2-e/kg lamb	Fossil energy MJ/kg lamb	Consumptive fresh water use L/kg lamb	Waterstress wsi eq L /kg lamb	Cultivated land use m^2/ kg lamb	Total land use m^2/ kg lamb
VIC CSF	16.5	28.5	149.8	22.3	3.0	56.7
VIC RAF	16.9	29.1	483.7	195.0	2.8	104.3
NSW CSF	17.3	22.9	142.4	58.1	4.7	81.2
NSW RAF	16.6	26.3	416.0	290.0	2.0	430.3
SA CSF	13.6	20.7	205.5	14.5	1.3	1363.0
SA RAF	15.5	31.0	517.1	27.3	3.1	277.2

Table 23 – Impact assessment per kg of greasy wool with biophysical allocation from the VIC, NSW and SA supply chains. CSF = case study farms, RAF = regional average farms

	GHG kg CO_2-e/kg wool	Fossil energy MJ/kg wool	Consumptive fresh water use L/kg wool	Waterstress wsi eq L /kg wool	Cultivated land use m^2/ kg wool	Total land use m^2/ kg wool
VIC CSF	19.9	16.8	188.7	20.6	2.5	76.1
VIC RAF	22.8	19.5	714.2	401.1	2.4	156.4
NSW CSF	22.0	10.7	185.8	65.6	6.4	115.6
NSW RAF	23.1	17.1	625.4	437.1	3.1	667.6
SA CSF	19.2	7.3	315.4	9.0	0.0	2239.5
SA RAF	21.1	22.8	777.7	207.0	2.9	426.6