Journal of Cleaner Production 94 (2015) 67–75

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Environmental impacts and resource use of Australian beef and lamb exported to the USA determined using life cycle assessment

Stephen Wiedemann [a,*], Eugene McGahan [a], Caoilinn Murphy [a], Ming-Jia Yan [a], Beverley Henry [b], Greg Thoma [c], Stewart Ledgard [d]

[a] FSA Consulting, PO Box 2175, Toowoomba, Qld 4350, Australia
[b] Institute for Future Environments, Queensland University of Technology, Brisbane, Qld 4000, Australia
[c] Ralph E. Martin Department of Chemical Engineering, University of Arkansas, Fayetteville, AR 72701, USA
[d] AgResearch Ruakura Research Centre, Private Bag 3123, Hamilton 3240, New Zealand

ARTICLE INFO

Article history:
Received 8 September 2014
Received in revised form
21 January 2015
Accepted 21 January 2015
Available online 31 January 2015

Keywords:
LCA
Water
GHG
Carbon
Energy
Footprint

ABSTRACT

Australia is one of the two largest exporting nations for beef and lamb in the world and the USA is a major export market for both products. To inform the Australian red meat industry regarding the environmental performance of exported food products, this study conducted the first multi-impact analysis of Australian red meat export supply chains including all stages through to warehousing in the USA. A large, integrated dataset based on case study farms and regional survey was used to model beef and lamb from major representative production regions in eastern Australia. Per kilogram of retail-ready red meat, fresh water consumption ranged from 441.7 to 597.6 L across the production systems, stress-weighted water use from 108.5 to 169.4 L H_2O-e, fossil energy from 28.1 to 46.6 MJ, crop land occupation from 2.5 to 29.9 m^2 and human edible protein conversion efficiency ranged from 7.9 to 0.3, with major differences observed between grass finished and grain finished production. GHG emissions excluding land use and direct land use change ranged from 16.1 to 27.2 kg CO_2-e per kilogram, and removals and emissions from land use and direct land use change ranged from −2.4 to 8.7 kg CO_2-e per kilogram of retail retail ready meat.

Process based life cycle assessment shows that environmental impacts and resource use were highest in the farm and feedlot phase. Transportation contributed ≤5% of greenhouse gas emissions, water and land, confirming that food miles is not a suitable indicator of environmental impacts for red meat transported by ocean shipping. The contribution of international transportation to total energy demand was higher, ranging from 14 to 23%. These beef and lamb supply chains were found to rely on small volumes of water from stressed water catchments, and occupied only small amounts of crop land suited to other food production systems. Production of high quality protein foods for human consumption used only small amounts of protein from human edible grain.

1. Introduction

Agricultural systems such as livestock production face the challenge of maintaining and increasing production in the future with constrained natural resources and pressure to reduce environmental impacts. Globally, meat demand is expected to increase 74% by 2050 because of expanding global population and increased wealth (FAO, 2009). However, global targets also exist to reduce greenhouse gas (GHG) emissions (Stocker et al., 2013) and concerns exist regarding the use of scarce water resources (Rockström et al., 2007; WHO, 2009) and arable land (UNEP, 2014). Australia is one of the two largest exporting nations for beef and lamb in the world, closely following Brazil in total volume of beef and New Zealand in lamb (FAO, 2011). The United States of America (USA) is a major export market for both products (DFAT, 2012). Product life cycle assessment (LCA) is an important method for understanding the impacts associated with food products and particularly for determining what stages in the supply chain contribute to impacts. Despite long transport distances, LCA studies of red meat have shown that transportation distance, or 'food miles' (Paxton, 2011) is not a good indicator of environmental impacts in several instances (Webb et al., 2013; Weber and Matthews, 2008). However, 'food

* Corresponding author. Tel.: +61 (0)7 4632 8230; fax: +61 (0)7 4632 8057.
E-mail address: stephen.wiedemann@fsaconsulting.net (S. Wiedemann).

http://dx.doi.org/10.1016/j.jclepro.2015.01.073

List of acronyms:

ABARES	Australian Bureau of Agricultural and Resource Economics and Sciences
CSF	case study farms
GHG	greenhouse gas
GWP	global warming potentials
HEP-CE	human edible protein conversion efficiency
LCA	life cycle assessment
LF	long fed
LU	Land Use
dLUC	direct land use change
MF	mid fed
NSW	New South Wales
QLD	Queensland
RAF	regional average farms
SA	South Australia
VIC	Victoria
WSI	water stress index

miles' is still taken as a proxy indicator for environmental impact in popular media communications and a greater understanding of the relationship between impacts and transport distance is sought by the users of Australian beef and lamb in the USA.

To date, there has been no holistic environmental analysis of Australian beef and lamb supply chains to the USA. Life cycle assessment studies of Australian production have focussed on case study farms (Eady et al., 2011; Peters et al., 2010a, 2010b), theoretical production systems (Ridoutt et al., 2012a) or controlled production systems found on research farms (Brock et al., 2013) that could not be considered representative of markets that draw from large production regions. These studies predominantly focussed on one or two impacts only. Recent farm gate studies of beef (Wiedemann et al., 2015c) and lamb (Wiedemann et al., 2015d) cover larger regions representative of Australia's export markets through to the farm gate, and were the basis for this expanded supply chain analysis. The present study aimed to determine major environmental impacts and resource use from the production, processing and transport of Australian beef and lamb to the USA by extending two existing farm-gate LCA studies by the same authors, which used large, integrated datasets based on case study farms and regional survey datasets. The study aimed to report on major environmental impacts and resource use indicators with new methods and to provide a robust assessment of impacts and hotspots in the supply chain, with particular attention to the role of transportation.

2. Materials and methods

The study included beef and lamb production from major representative production regions in eastern Australia, through the whole supply chain to the point of distribution to retail in the USA. The functional unit was chosen as one kilogram of retail ready cuts of Australian beef and lamb, at the regional storage centre in the USA. The system boundary included all stages of production, processing, transport and cold storage on the east coast of the USA, as well as distribution to the point of retail (Fig 1).

2.1. Production system characteristics

Australia's sheep and cattle industries have been developed to utilise some 3.5 M km^2 of native vegetation grazing land (Leslie and Mewett, 2013), or 46% of continental land area. The majority of sheep are produced in the south-east states of South Australia (SA), Victoria (VIC) and New South Wales (NSW), representing 73% of Australia's sheep flock (MLA, 2013a) and the vast majority of export lamb production. The majority of beef cattle are produced in the states of Queensland (QLD) and NSW. Central and southern QLD, and northern and central NSW represent 35% of Australia's beef herd and the major regions exporting premium beef to the USA market and were the focus of the study. Premium beef and lamb exports to the USA must meet specific market requirements. Export lambs are >22 kg carcase weight (CW) and beef cattle destined for premium markets may be grass-fed or grain-finished. The study investigated beef bred in rangeland areas and finished on pastures (grass-fed), and steers finished on grain for either 115 days (Mid-fed – MF) or 330 days (Long-fed – LF). The LF category is tailored to the production of a high quality, niche beef product, predominantly from Angus or Wagyu breeds, for the USA restaurant trade.

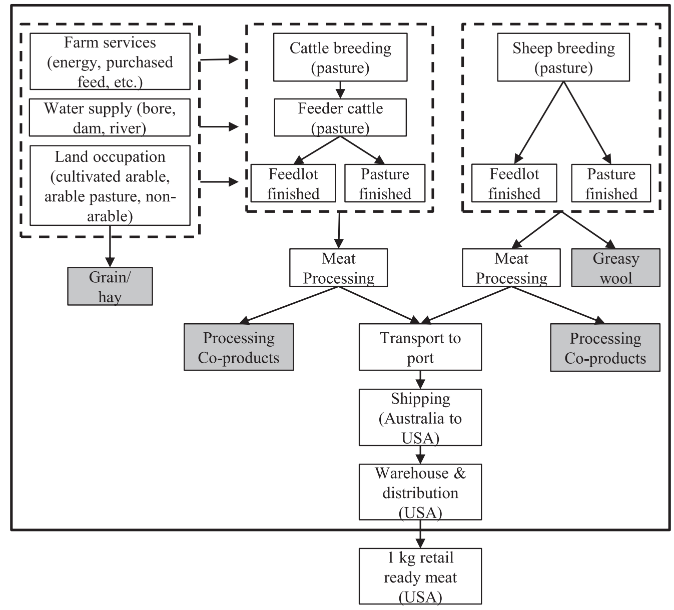

Fig. 1. Illustration of beef and lamb supply chains in the study. Shaded boxes indicate co products.

2.1.1. Indicators

The study investigated GHG emissions using the IPCC AR4 global warming potentials (GWP, 100 years) of 25 for methane and 298 for nitrous oxide (IPCC, 2007). Greenhouse gas emissions associated with land use (LU) and direct land use change (dLUC) were included and reported separately.

Fossil fuel energy demand was assessed by aggregating all fossil fuel energy inputs throughout the system and reporting these per mega joule (MJ) of energy, using Lower Heating Values (LHV). Modelling methods and processes used are described below. Fresh water consumption (an inventory method – Bayart et al., 2010) was assessed, covering all sources and losses associated with livestock production in both foreground and background systems. Fresh water consumption refers to evaporative uses or uses that incorporate water into a product that is not subsequently released back into the same river catchment (ISO, 2014). Stress-weighted water use was assessed using the water stress index (WSI) of Pfister et al. (2009) reported in water equivalents (L H_2O-e) after Ridoutt and Pfister (2010). Land occupation was reported using a disaggregated inventory based on land type and suitability. Four land types are specified, crop land, arable pasture land, non-arable pasture land and industrial land occupation. No characterisation factors were applied, and land occupation data were reported in square metre years (m^2 yr). Human edible protein conversion efficiency was determined by dividing the protein content of the retail ready product by human edible protein in the feed consumed and methods are explained in more detail in the following sections.

2.2. Life cycle inventory and modelling

The inventory stage was divided into segments covering the production phase in Australia, the meat processing stage in Australia, and transport and warehousing through the supply chain to the USA. Data collection and modelling methods are outlined for each stage.

2.2.1. Farm and feedlot

Inventory data covering livestock production, inputs of purchased feed, fertiliser, fuel and services, and land occupation for the farm stage of the supply chain were from regional survey data collected by the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) and a survey of case study farms (CSF). The ABARES dataset covered 345 beef producers and 203 specialist lamb producers annually for the five year period of 2006–2010 (ABARES, 2013a, b). Case study farm data were sourced from other publications (Wiedemann et al., 2015c, 2015d). Feedlot inventory data were from a survey of the Australian feedlot industry (Davis et al., 2010a, b). In addition to purchased inputs, Davis et al. (2010a) provided an inventory of measured water use and feed requirements for feedlot finishing. Material inputs for the farm and feedlot are provided in the supplementary material. For each region, human edible protein consumed by livestock was determined from the inventory of grain use throughout the supply chain based on the mass of human edible grain for each grain type, and the average protein content of the grain. Impacts associated with farm infrastructure were excluded based on the findings of a scoping study showing the contribution from these sources was <1% (Wiedemann, unpublished data). Inventory data were aggregated for reporting relative to livestock feed intake.

For each major region, a regional average farm model was developed based on the herd and flock inventory data, which determined total production outputs from the farm stage. This model was also used to determine total feed requirements (dry matter intake – DMI) for the grazing herds and flocks in each region, using the prediction equation of Minson and McDonald (1987) for cattle, and the AFRC (1990) feed intake model for sheep as they are applied in the Australian NGGI (DCCEE, 2012a). Results from the regional herds and flocks were aggregated to provide a market average for the USA export market with ratios for beef of 65% from NSW and 35% from QLD for the grass-fed and mid-fed grain finished supply chains. The long-fed supply chain was only supplied by NSW where a greater number of suitable cattle are bred. Lamb exports were modelled with equal proportions of lamb from SA, VIC and NSW. Herd and flock modelling methods are explained in detail in other publications (Wiedemann et al., 2015c, 2015d) and key herd and flock parameters are provided in Table 1. The herd and flock model also predicted livestock GHG emissions using methods from the Australian NGGI (DCCEE, 2012a) or global inventory methods (Dong et al., 2006) as applied in other publications (Wiedemann et al., 2015c, 2015d). Key model parameters are specified in Table 2. Emissions and removals from Land Use and direct Land Use Change were modelled for each region using methods outlined in previous related publications (Henry et al., Submitted for publication; Wiedemann et al., 2015c, 2015d). These methods are outlined in the supplementary material.

Table 1
Animal productivity characteristics of the grass-fed beef and lamb supply chains on a per farm basis at the farm gate.

	Grass-fed beef	Lamb
Breeding animals (no. cows/ewes)	222.8	1245
Breeding animal culling rate (%)	16.8	23
Breeding animal mortality rate (%)	2.1	4
Breeding animal average weight[a] (kg LW/head)	469	65
Weaning percent (%)	81.1	97
Weaning weight (kg LW)	208.7	n.r.
Average steer weight at sale (kg LW/head)	554.2	n.a
Average lamb weight at sale (kg LW/head)	n.a	51
Average daily growth rate of steers/lambs (kg LW/head/day)	0.53	0.14
Annual cattle sales (kg LW)	85,889	n.a.
Annual sheep and lamb sales (kg LW)	n.a.	63,965
Annual wool sales (kg greasy)	n.a	8284
Biophysical allocation for sheep LW (%)	n.a.	71

[a] LW = live weight, n.a. = not available, n.r. = not reported.

Each regional farm model estimated livestock drinking water based on livestock numbers, feed intake and climate. Farm water supply losses and irrigation water use were determined for each region using water balance methods and datasets described in detail by Wiedemann et al. (2015a, 2015b). For each regional farm, a disaggregated model of land occupation was developed to differentiate between crop land, arable pasture land and non-arable pasture land, using methods described in other publications Wiedemann et al. (2015a, 2015b).

Impacts associated with purchased inputs such as fertiliser and fuel were modelled using processes from the Australian LCI database (Life Cycle Strategies, 2007) where available, or the European Ecoinvent (2.0) database (Frischknecht et al., 2005). Feed grain inventory data were obtained from Wiedemann et al. (2010a) and Wiedemann and McGahan (2011). Impacts associated with services such as communications, insurance and accounting were modelled based on expenditure using economic input–output data (Rebitzer et al., 2002).

2.2.2. Meat processing

Inputs and impacts associated with beef and lamb processing (Table 3) were collected from an industry survey of meat processing plants in Australia (GHD, 2011) which included recorded energy inputs, cleaning inputs, water use and waste water production per unit of output.

Two meat processing plant models were developed to determine flows of co-products and waste from processing of beef and lamb, and to determine emissions from waste treatment. Factors used to determine product mass, co-product mass and waste production are described in Wiedemann and Yan (2014). Key model parameters included dressing percentages (i.e. from live animal to hot carcase) of 55% and 45%, cutting and chilling losses of 3% and 4%, and retail yield (from cold carcase to retail meat) 74% and 88% for boneless beef and bone-in lamb respectively. Human edible protein conversion efficiency was determined by dividing the mass of protein in boneless beef and bone-in lamb, determined from the edible yield of each product (Wiedemann and Yan, 2014) and a dry mass protein content of 0.2 kg/kg red meat. Protein mass was therefore 0.19 and 0.15 kg/kg retail product for beef and lamb respectively, and these values were used to determine by the total human edible protein yield. Emissions from anaerobic treatment of effluent were included using effluent production rates and characteristics reported by GHD (2011) and from data collected at three meat processing plants by the authors, and Australian inventory factors (DCCEE, 2012a).

Table 2
Greenhouse gas (GHG) methods and uncertainty estimates used in the study.

Emission source	Key parameters/model	Uncertainty (%)	Reference
Enteric methane[a] (kg/sheep)	kg DMI/head × 0.0188 + 0.00158	±20	DCCEE (2012a).
Enteric methane (CH_4 yield – % of GEI[b] grazing cattle)	6.5%	±20	Dong et al. (2006).
Enteric methane[c] (MJ CH_4 feedlot cattle)	3.406 + 0.510*SR + 1.736*H + 2.648*C	±20	DCCEE (2012a).
Manure methane (grazing) (kg)[d]	kg DMI* (1 – DMD)* MEF	±20	DCCEE (2012a).
Manure nitrous oxide (grazing)	Urinary N – 0.004 kg N_2O–N/kg N in urine Faecal N – 0.005 kg N_2O–N/kg N in faeces.	±50	DCCEE (2012a).
Manure ammonia (grazing)	0.2 kg NH_3–N/kg N of excreted in manure	±20	DCCEE (2012a).
Manure methane (kg/hd feedlot beef)[e]	VS (kg/head)* Bo* MCF* p	±20%	DCCEE (2012a).
Manure nitrous oxide (feedlot cattle)	Faecal and urinary N – 0.01 kg N_2ON/kg N excreted	±50%	Muir (2011).
Manure ammonia (feedlot cattle)	0.75 kg NH_3–N/kg N excreted	±20%	Watts et al. (2012).
Indirect nitrous oxide – ammonia	0.01 kg N_2O–N/kg N lost as ammonia-N	±50	DCCEE (2012a).
Indirect nitrous oxide – Leaching and runoff	0.0125 kg N_2O–N/kg N lost in leaching and runoff	±50	DCCEE (2012a).

[a] DMI = Dry Matter intake.
[b] GEI = Gross energy intake assumed to be 18.6 MJ/kg dry matter intake (DMI).
[c] SR = soluble residues, H = hemicellulose, C = cellulose.
[d] DMD = Dry matter digestibility; MEF = Methane emission factors (temperate – 1.4×10^{-5} kg CH_4/kg DM manure, warm – 5.4×10^{-5} kg CH_4/kg DM manure).
[e] VS = volatile solids; Bo = 0.17 m^3 CH_4/kg VS; MCF = 0.05 in Queensland, 0.015 in NSW. Density of methane (p) = 0.622 kg/m.[3].

2.2.3. Transport and warehousing

Transport stages were included throughout the supply chain based on an inventory of transport distances and representative truck types and load specifications (Table 4). International transport of chilled beef and lamb was via ship to the USA. Transport distances represented export from the port of Brisbane and import to the port of Philadelphia (PH) based on trade data (https://usatrade.census.gov/). Alternative modelling to the port of Los Angeles was completed for comparison. An inventory of inputs associated with storage in refrigerated warehouse was based on micro data from the Energy Information Agency Commercial Buildings Energy Consumption Survey, ASHRAE design guidelines (ASHRAE, 2004; EIA, 2003), and an interview of plant managers to determine electricity consumption, conducted as part of the project (Table 4).

Impacts associated with transport were modelled for each different transportation type using database processes corresponding to the specific inventory stage. Australian truck processes were modelled using the AustLCI database (Life Cycle Strategies, 2007). International ocean liner transport was modelled using data from Webb et al. (2013). GHG emissions from refrigerant losses were not included because of a lack of data and their expected minor contribution (Webb et al., 2013). Ships were assumed to carry goods both ways because Australia operates a large trade deficit with the USA (DFAT, 2012). For transport within the USA, US lifecycle inventory (National Renewable Energy Laboratory, 2012) long-haul truck processes were applied. To understand the sensitivity of modelling assumptions in transportation, baseline parameters (Table 4) used to model the transportation from Australia to the USA and warehousing in USA were varied as part of the sensitivity analysis.

2.3. Methods for handling co-production

The study applied allocation methods following the (ISO, 2006) hierarchy and guidance from (LEAP, 2014) where clear and suitable methods were available. System subdivision and allocation prior to the farm-gate was explained in Wiedemann et al. (2015c, 2015d), using a system separation where possible, and biophysical allocation between wool and live weight in the sheep system after Wiedemann et al. (2015b).

At the meat processing plant, raw hides, tallow, meat and blood meal are co-generated. Allocation between meat products and co products at the point of meat processing was handled using economic allocation (LEAP, 2014) based on Australian market data (MLA, 2013b, c) and allocation fractions are provided in Table 5.

To understand the sensitivity of the co-product handling at meat processing, an alternative, hybrid approach was analysed. In this approach, we divided the rendering process from the rest of meat processing system. Biophysical allocation was used between retail meat, edible offal and raw hides, and system expansion was applied the rendering products (pet food, protein meals, and tallow). Production of soybean meal (equally sourced from Brazil and Argentina) was substituted for production of pet food, and protein meals. Tallow was substituted for Malaysian palm oil. The hybrid approach was chosen as an alternative to more closely match the biophysical approach taken at the farm stage where wool and live weight allocation was performed for sheep. A detailed explanation of meat processing allocation and system expansion factors and methods is described in Wiedemann and Yan (2014).

2.4. Analysis

Modelling and the uncertainty analysis was conducted in Simapro 7.3 (Pré-Consultants, 2012). Uncertainty associated with purchased inputs was determined using a pedigree matrix (Frischknecht et al., 2005), and was assessed using Monte Carlo analysis in SimaPro 7.3. One thousand iterations provided a 95% confidence interval for the results. Differences between datasets were assessed using comparative Monte Carlo analysis in SimaPro 7.3.

3. Results

Results are presented for resource use and impact categories of retail ready beef and lamb in the following sections.

Table 3
Major inputs associated with meat processing used in this study.

Major inputs	units	Per tonne carcase weight (beef)	Per tonne carcase weight (lamb)
Fresh water consumption (100% consumptive)	L	8743	5973
Electricity	kWh	318	329
LPG	MJ	83	533
Diesel	MJ	40	19
Petrol	MJ	7	14
Coal	MJ	693	
Fuel oil	MJ		1184
Natural Gas	MJ	1230	2346

Table 4
Assumptions and emission factors used for transportation modelling from meat processing plant to regional storage centre.

	Units	Value	References
Baseline parameters			
Distance from meat processing plant to port	km	350	Based on average distances from Rockhampton, Kilcoy, Dinmore, Oakey, Inverell, Tamworth to Brisbane port
Distance form custom clear to warehouse	km	30	Expert judgement based on a review of meat import processes and interviews with importers
Warehousing period	m^{3}*yr	0.08	Expert judgement based on a review of meat import processes and interviews with importers
Distance from warehouse to regional retail centre	km	600	Expert judgement based on a review of meat import processes and interviews with importers
Shipping distance from Australian port to USA	km	18120	Oceanic distance between Brisbane and Philadelphia
Emission factors			
Transoceanic shipping			
Energy	MJ/tkm	0.2–0.27	Webb et al. (2013).
GHG emissions	kg CO_2 e/tkm	0.02–0.0215	Webb et al. (2013).
Warehousing			
Electricity	kWh/(m^3,yr)	30.4–512.8	(ASHRAE, 2004; EIA, 2003)
Natural gas	Btu/(m^3,yr)	21019–535820	(ASHRAE, 2004; EIA, 2003)
Ammonia	kg/(m^3,yr)	0–7.7	(ASHRAE, 2004; EIA, 2003)

3.1. Fresh water consumption and stress-weighted water use

Fresh water consumption ranged from 441.7 to 597.6 L/kg boneless beef depending on the production system and averaged 463.8 L/kg bone-in lamb (Fig 2). Drinking water and irrigation were the largest sources of fresh water consumption. With supply losses included, drinking water supply contributed 63 and 56% of fresh water consumption, and irrigation contributed 31 and 61% for the average of the beef and lamb supply chains respectively. Water supply losses were much higher than livestock drinking water due to the large evaporative loss from farm dams, which were the major drinking water supply. Irrigation losses were lower than irrigation water use due to the major supply source being larger, more efficient supply dams or bores. Stress-weighted water use ranged from 108.5 to 124.9 L-e/kg boneless beef and averaged 169.4 L-e/kg bone-in lamb. Stress-weighted water use was strongly influenced by regional water stress indexes, which averaged 0.22 (0.02–0.85) and 0.37 (0.01–0.82) in the beef and lamb production regions. This resulted in higher stress-weighted water use for lamb than beef, despite the higher average fresh water consumption for the latter.

3.2. Land occupation

Total land occupation varied from 199.4 to 432.5 m^2/kg boneless beef, 88% of which was from non-arable land (Fig 2). Crop land occupation from the grass and grain finished beef supply chains varied from 3.2 to 29.9 m^2/kg boneless beef, with the small amount of crop land used for grass-fed beef contributed by forage or supplement production. Grain production was the predominant contributor to crop land occupation. Consequently, Beef MF and LF occupied 5.1 to 9.5 times more crop land than grass finished beef. Crop land occupation from the lamb supply chain averaged 2.5 m^2/kg bone-in lamb.

Table 5
Meat products and co products per 1000 kg of live weight of beef and lamb processed.

Products	Beef		Lamb	
	Mass of product (kg)	Economic allocation	Mass of product (kg)	Economic allocation
Retail cuts[a]	442.1	91.4%	426.7	88.4%
Edible portion of retail cuts[b]	95%		76%	
Hides	88.4	5.5%	75.0	7.2%
Meat, blood and bone meal	61.9	1.5%	69.8	1.6
Tallow	33.6	1.1%	81.0	2.3
Pet food	13.3	0.5%	15.0	0.5
Totals	639.3	100%	667.5	100%

[a] For beef, retail cuts refer to boneless beef and edible offal; for lamb, this refers to bone-in lamb and edible offal.
[b] Wiedemann and Yan (2014).

3.3. Fossil fuel energy demand

Total fossil fuel demand from the grass and grain finished beef supply chains ranged from 32.3 to 46.6 MJ/kg boneless beef (Fig 2). The largest contribution was from the farm and feedlot stage (averaging 63%), followed by meat processing (averaging 20%), and international transportation (predominantly shipping) of meat from Australia to the warehouse in the USA (averaging 17%). Fossil fuel demand associated with transportation in the production system contributed a small amount (1–2%) of total energy demand. Grain-finished supply chains (Beef MF and LF) were found to have higher energy intensity than the grassland-finished supply chain. Total fossil fuel energy demand for the lamb supply chain averaged 28.1 MJ/kg bone-in lamb (Fig 2). The largest contribution was from the farm production stage (46%), followed by meat processing (31%) and international transportation (23%).

3.4. Greenhouse gas emissions

Emissions (excl. LU, dLUC) from the grass and grain finished beef supply chains ranged from 23.4 to 27.2 kg CO_2-e/kg beef. The predominant contribution was from primary production (averaging 93%), followed by meat processing (4%) and transportation of meat from Australia to the warehouse in the USA (3%). By source, enteric methane was the single largest emission (averaging 70%), followed by carbon dioxide from the burning of fossil fuels (averaging 11%), and manure nitrous oxide (averaging 10%). Total GHG emissions from lamb supply chain averaged 16.1 kg CO_2-e/kg bone-in lamb. Relative contributions of components were similar between the beef and lamb supply chains. Lamb primary production was also the predominant contributor (90%) and 75% of emissions were from enteric methane. Contributions from carbon dioxide were slightly higher (14%) while manure nitrous oxide (9%) was slightly lower for lamb GHG emissions relative to beef emissions.

Mean LU and dLUC emissions from the low and high emission scenarios were 8.3, 8.7 and 4.1 kg CO_2-e/kg boneless beef for grass-

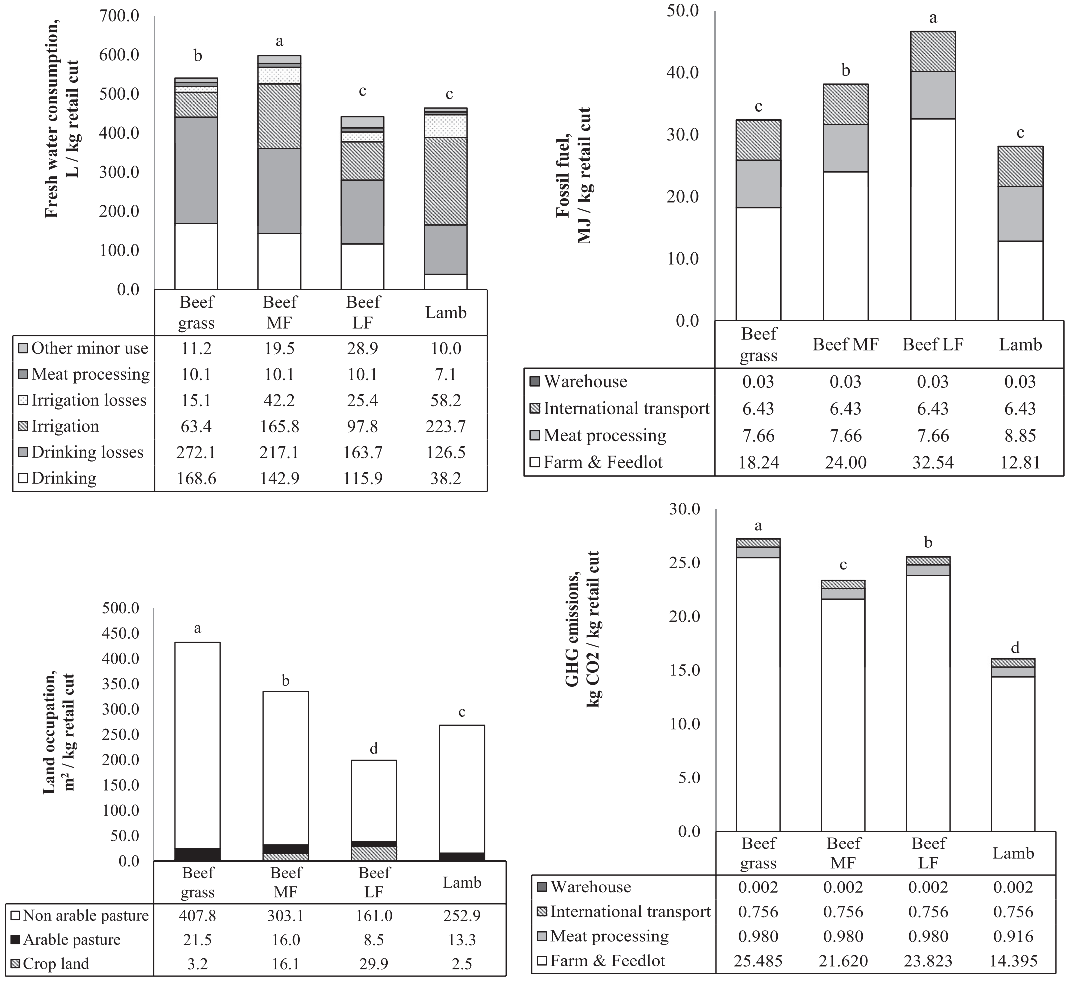

Fig. 2. Contribution of processes to fossil energy, fresh water consumption, land occupation, and greenhouse gas (GHG, excluding land use and direct land use change) emissions per kg of retail ready Australian beef and lamb exported to USA. Grass = grass-fed, MF = medium fed grain (115 days), LF = long-fed grain (330 days). Different letters on bars indicate significant differences (P < 0.05) between cases assessed using comparative Monte Carlo analysis.

fed, mid-fed and long-fed grain finished beef respectively when soil organic carbon stock change was assumed to be zero under pasture. When soil carbon sequestration was included for pastures for beef produced in NSW, values declined slightly to 7.3, 8.1 and 3.5 kg CO_2-e/kg boneless beef for grass-fed, mid-fed and long-fed beef respectively. Results from the two scenarios for lamb showed average emissions of 0.4 kg CO_2-e/kg bone-in lamb, or removals of −2.4 kg CO_2-e/kg lamb where soil carbon sequestration under improved pastures was included.

3.5. Human edible protein conversion efficiency

Total HEP-CE was 7.9 for grass finished boneless beef and 2.9 for bone-in lamb, indicating that these products yield more human edible protein than they utilise throughout the system via consumption of animal feeds that could potentially be fed to humans. Grain finished boneless beef utilised more human edible protein inputs, resulting in lower efficiencies of 0.3 (LF) and 0.5 (MF).

4. Discussion

Livestock production, processing and transport requires water and energy inputs, and utilises land resources. While a number of studies have investigated resource use at the farm-gate level (Ridoutt et al., 2012a; Wiedemann et al., 2015c, 2015d; Williams et al., 2006), fewer have included a detailed analysis of meat processing and the associated allocation processes required. We found the primary production phase of the supply chain to dominate GHG emissions, fresh water consumption and land occupation, while fossil fuel energy demand was more evenly distributed across the supply chain.

The farm-gate component of this study extended the assessment of fresh water consumption by taking into account consumptive losses associated with water supply throughout the supply chain, accounting for more loss pathways than previously included by some authors (e.g. Peters et al., 2010b). Farm-gate water use was slightly lower than reported for the national average for Australian beef (Wiedemann et al., 2015a), reflecting differences between the specific regions supplying the USA prime beef market and a broader assessment of Australian beef production. Several published studies from the United States do not report their coverage of the supply chain, but report higher water use of ~2300 L/kg boneless beef (converted from carcase weight – Capper, 2011) to 3682 L/kg boneless beef (Beckett and Oltjen, 1993). Comparative stress weighed water use was found to be much lower than fresh water consumption, reflecting the relatively low water stress conditions existing in most Australian livestock producing regions. Similar findings were made for Australian lamb by Ridoutt et al. (2012b).

We found Australian livestock production to rely predominantly on non-arable rangelands, reflecting the land capability in

Australia, where less than 6.5% of land mass is considered arable (FAOSTAT, 2014). While total land occupation in the study was higher than found by others (e.g. Nguyen et al., 2010; Williams et al., 2006), comparisons are not meaningful without understanding land capability or disturbance. Further research investigating biodiversity impacts is required to determine impacts from land occupation more accurately, but at a minimum differentiating the use of crop land (high disturbance) from pasture land (low disturbance) is informative. Production of the major grass-finished beef and lamb exports was found to utilise very little grain, yielding more HEP than consumed by the animals. This study found energy intensity per kilogram of retail ready meat to be lower from grass finishing than grain finishing. The higher energy intensity from grain finishing primarily relates to the additional inputs required to produce, transport and mill the feed inputs. In comparison, the grass finishing systems used few inputs for pasture production. Together with the land occupation results, this finding shows the importance of ruminant livestock in producing food products from low value land and grasses unsuitable for alternative food production. This is particularly relevant for Australia, where the area of arable land is small comparative to the area of natural rangelands.

Greenhouse gas emissions are known to be dominated by the production phase of the supply chain in ruminants (Ledgard et al., 2011). Results from the production phase were similar to previous Australian studies of beef (Eady et al., 2011; Peters et al., 2010a) and lamb (Brock et al., 2013; Eady et al., 2012) when GWP values and enteric methane prediction models were standardised. Impacts from lamb were lower than beef at the farm gate predominantly because a proportion of impacts are allocated to wool (see supplementary material) and because the sheep systems had higher productivity, resulting in lower livestock emissions. Emissions from Australian beef excluding LU and dLUC were similar or slightly lower than results from suckler beef production in Europe (e.g. Casey and Holden, 2006; Nguyen et al., 2012; Williams et al., 2006) and North America (e.g. Beauchemin et al., 2010; Lupo et al., 2013; Pelletier et al., 2010). This is mainly due to lower energy inputs on Australian cattle farms, and lower nitrous oxide emissions from cropping and manure in Australia than international defaults (DCCEE, 2012a). Differences in the impacts from beef and lamb diminished when the different yield of edible product (Table 5) was taken into account; per kg of edible product, GHG emission intensities ranged from 24.5 to 28.6 per kg beef and 21.1 per kg lamb.

Fewer studies have assessed impacts from LU and dLUC sources though these may be significant. Estimation and attribution of emissions and removals is complex, and results were regionally variable, contributing 4.1–8.7 kg CO_2-e/kg beef. Removals or modest emission levels from the sheep system ranged from −2.4 to 0.4 kg CO_2-e/kg lamb, in response to modest rates of soil carbon sequestration under pastures in southern Australia. Emissions from dLUC have declined rapidly in Australia (DCCEE, 2012b) in response to changed management practices and legislation introduced over the last decade. These changes are only realised slowly when applying a retrospective analysis and 20-year amortization. We explored the expected future emissions from Australian beef by projecting the clearing rate data forward over the period from 2006 to 2026, when the full impact of reduced deforestation will take effect. This showed that dLUC emissions for Australian beef will decline to between 0.4 and 0.7 kg CO_2-e/kg beef, with potential for reforestation on previously cleared land to result in net removal of carbon dioxide in this time period (Henry et al. submitted). Rates of GHG removals associated with increased soil carbon levels in improved pasture for the lamb production system are expected to decline as a new equilibrium is reached. Reported dLUC in the literature range from 5 to 8 kg CO_2-e/kg beef for dairy calves produced in the EU (Nguyen et al., 2010) to >700 kg CO_2-e/kg beef in Brazil (Cederberg et al., 2011). In contrast to these results, Pelletier et al. (2010) investigated soil carbon sequestration in grazing systems and suggested that removals may be up to 8 kg CO_2-e/kg beef for cattle raised on improved pastures in the USA, though a complete investigation of LU and dLUC emissions and removals was not included. Considering the limited data within the LCA literature, further research is required to understand the role of LU and dLUC on beef production.

4.1. Handling co-production

Processing and transport are minor stages and are occasionally overlooked in studies of beef and lamb. Results from a number of recent and older studies (e.g. Lupo et al., 2013; Williams et al., 2006) report results using a carcase weight functional unit without including impacts or allocation processes involved in meat processing, showing a mismatch between the system boundary and reference flow or functional unit of the study (Wiedemann and Yan, 2014). By accounting for the value of co-products and impacts from the meat processing stage, impacts to grass-fed beef, for example, were in the order of 10% lower than if co-products were ignored in the present study. To understand the sensitivity of economic allocation at the meat processing, a hybrid approach was also performed to divide impacts between the primary meat products while using system expansion to account for the minor rendering products. While it has been accepted for some time in LCA practice that multiple methods of allocation may be used at different points in the same study, here this was applied for a closely related co-product system coming from the same process. The hybrid approach resulted in an average of 8% lower GHG emissions, fossil fuel use, fresh water consumption, and stress-weighted water use, and 21% lower crop land occupation for the primary products from beef and lamb supply chains. This highlighted the sensitivity of allocation at meat processing and the contribution of valuable co products from meat processing.

4.2. Transportation

When averaged across the beef and lamb supply chains, international transportation contributed contributed ≤5% to GHG emissions, land occupation and fresh water consumption, while the contribution to energy demand was higher, ranging from 14% to 20% for beef and 23% for lamb. The higher relative contribution for lamb was related to the lower energy intensity at farm gate (Fig 2). To test assumptions, we investigated alternative transport distances to port and within the USA. Importing beef and lamb into the closer port (Los Angeles) reduced GHG by 0.3% and a modest impact on energy use (−4%). Increasing the distance of post-warehouse transport from 600 km to 2000 km increased the overall energy use by 6–7%, while changes to GHG were ~1%. The minor role of transport in assessing impacts from globally traded red meat has been shown previously (Ledgard et al., 2011; Webb et al., 2013) where transport contributed similar levels to GHG and energy as found here. Thus, transport distance or 'food miles' was found to be a poor predictor of the environmental impacts and resources of red meat transported by ship.

5. Conclusions

This study is the first multi-impact analysis of Australian red meat supply chains through to the USA. Impacts and resource use associated with red meat supply are heavily influenced by the production system and less by other components of the supply chain such as transportation or meat processing. Transportation was found to contribute ≤5% of GHG emissions, and water and land resource use, confirming that food miles is not a suitable indicator of environmental impacts for red meat. Emissions from livestock, soil and fossil fuel consumption were similar in aggregate to other production regions in the world, and LU and dLUC emissions were found to be a moderate emission source for beef, but not lamb, in the current analysis. Deforestation emissions have declined significantly for Australian beef and as a consequence, the emissions attributed to beef from historic clearing events will continue to decline. Reforestation and soil carbon sequestration were shown to have potential to represent a small offset for emissions from beef and lamb production. Water use was found to be relatively low and was drawn predominantly from low water stress catchments. In the first Australian analysis of regional land occupation and consumption of human edible protein inputs, red meat products were found to be produced predominantly from non-arable rangeland areas with small amounts of arable land occupation. Human edible grain inputs were modest for the grass finished systems, displaying a high degree of resource efficiency in the production of high quality food from low quality grass resources, with a low degree of competition with human food sources or other grain users. These results underscore the valuable role in the global food supply chain that ruminants can provide when managed to exploit marginal land which is poorly suited for arable production.

Acknowledgements

This study was funded by Meat and Livestock Australia, project number B.CCH.2072. Review and input to the project scope from Dr. Tom Davison was appreciated. The contribution of data from the many case study farmers involved in the study was greatly appreciated.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2015.01.073.

References

ABARES, 2013a. All Beef Industries Combined, Australian Agricultural and Grazing Industries Survey (AAGIS). Australian Bureau of Agricultural and Resource Economics and Sciences. Department of Agriculture, Fisheries and Forestry, Canberra, ACT.

ABARES, 2013b. Slaughter Lambs with More than 20% Receipts from Prime Lamb Sales, Australian Agricultural and Grazing Industries Survey (AAGIS). Australian Bureau of Agricultural and Resource Economics and Sciences, Department of Agriculture, Fisheries and Forestry, Canberra, ACT.

AFRC, 1990. Nutritive Requirements of Ruminant Animals: Energy. CAB International.

ASHRAE, 2004. Advanced Energy Design Guides. Viewed October 2013. https://www.ashrae.org/standards-research–technology/advanced-energy-design-guides.

Bayart, J.-B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., Koehler, A., 2010. A framework for assessing off-stream freshwater use in LCA. Int. J. Life Cycle Assess. 15, 439–453.

Beauchemin, K.A., Henry Janzen, H., Little, S.M., McAllister, T.A., McGinn, S.M., 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: a case study. Agric. Syst. 103, 371–379.

Beckett, J.L., Oltjen, J.W., 1993. Estimation of the water requirement for beef production in the United States. J. Anim. Sci. 71, 818–826.

Brock, P.M., Graham, P., Madden, P., Alcock, D.J., 2013. Greenhouse gas emissions profile for 1 kg of wool produced in the Yass Region, New South Wales: a Life Cycle Assessment approach. Anim. Prod. Sci. 53, 495–508.

Capper, J., 2011. The environmental impact of beef production in the United States: 1977 compared with 2007. J. anim. Sci. 89, 4249–4261.

Casey, J.W., Holden, N.M., 2006. Greenhouse gas emissions from conventional, agri-environmental scheme, and organic Irish suckler-beef units. J. Environ. Qual. 35, 231–239.

Cederberg, C., Persson, U.M., Neovius, K., Molander, S., Clift, R., 2011. Including carbon emissions from deforestation in the carbon footprint of brazilian beef. Environ. Sci. Technol. 45, 1773–1779.

Davis, R., Wiedemann, S., Watts, P., 2010a. Quantifying the Water and Energy Usage of Individual Activities Within Australian Feedlots -Part a Report: Water Usage at Australian Feedlots 2007–2009. Meat & Livestock Australia Limited, North Sydney, NSW.

Davis, R., Wiedemann, S., Watts, P., 2010b. Quantifying the Water and Energy Usage of Individual Activities Within Australian Feedlots -Part B Report: Energy Usage at Australian Feedlots 2007–2009. Meat & Livestock Australia Limited, North Sydney, NSW.

DCCEE, 2012a. National Inventory Report 2010, Volume 1. The Australian Government Submission to the United Nations Framework Convention on Climate Change, Australian National Greenhouse Accounts. Department of Climate Change and Energy Efficiency, Canberra, Australia.

DCCEE, 2012b. National Inventory Report 2010, Volume 2. Land Use, Land Use Change and Forestry, Australian National Greenhouse Accounts. Department of Climate Change and Energy Efficiency, Canberra, Australia.

DFAT, 2012. Australia's Trade with the Americas. Trade Advocacy and Statistics Section. Department of Foreign Affairs and Trade, Australia.

Dong, H., Mangino, J., McAllister, T.A., Hatfield, J.L., Johnson, D.E., Lassey, K.R., Aparecida de Lima, M., Romanovskaya, A., Bartram, D., Gibb, D.J., Martin, J.H.J., 2006. Emissions from livestock and manure management. In: Eggleston, S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global Environmental Strategies, Kanagawa, Japan.

Eady, S., Carre, A., Grant, T., 2012. Life cycle assessment modelling of complex agricultural systems with multiple food and fibre co-products. J. Clean. Prod. 28, 143–149.

Eady, S., Viner, J., MacDonnell, J., 2011. On-farm greenhouse gas emissions and water use: case studies in the Queensland beef industry. Anim. Prod. Sci. 51, 667–681.

EIA, 2003. Commercial Buildings Energy Consumption Survey. U.S. Energy Infomation Administration. Viewed October 2013. http://www.eia.gov/consumption/commercial/data/2003/index.cfm?view=consumption#c1.

FAO, 2009. How to Feed the World in 2050. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2011. IMPORTS: Commodities by Country, FAOSTAT. Food and Agriculture Organisation of the United Nations.

FAOSTAT, 2014. Food and Agriculture Organisation of the United Nations: Statistics. FAO, Rome, Italy.

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2005. The ecoinvent database: overview and methodological framework. Int. J. Life Cycle Assess. 10, 3–9.

GHD, 2011. Industry Environmental Sustainability Review 2010 (Sydney).

Henry, B.K., Butler, D., Wiedemann, S., 2015. A life cycle assessment approach to quantifying greenhouse gas emissions from land use change for beef production in eastern Australia. Rangel. J. (Submitted for publication).

IPCC, 2007. Climate Change 2007: the Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Revised ed. Cambridge University Press, Cambridge, United Kingdom and New York, USA.

ISO, 2006. Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organisation for Standardisation, Geneva, Switzerland.

ISO, 2014. Environmental Management – Water Footprint – Principles, Requirements and Guidelines. ISO 14046:2014. International Organisation for Standardisation, Geneva, Switzerland.

LEAP, 2014. Greenhouse Gas Emissions and Fossil Energy Demand from Small Ruminant Supply Chains: Guidelines for Quantification. Livestock Environmental Assessment and Performance Partnership. FAO, Rome, Italy.

Ledgard, S.F., Lieffering, M., Coup, D., O'Brien, B., 2011. Carbon footprinting of New Zealand lamb from the perspective of an exporting nation. Anim. Front. 1, 27–32.

Lesslie, R., Mewett, J., 2013. Land Use and Management: the Australian Context. Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES).

Life Cycle Strategies, 2007. Australian Unit Process LCI Library and Methods. Version 2009. 11. November 2009.

Lupo, C.D., Clay, D.E., Benning, J.L., Stone, J.J., 2013. Life-cycle assessment of the beef cattle production system for the northern great Plains, USA. J. Environ. Qual. 42, 1386–1394.

Minson, D.J., McDonald, C.K., 1987. Estimating forage intake from the growth of beef cattle. Trop. Grassl. 21, 116–122.

MLA, 2013a. Fast Facts. Australia's Sheepmeat Industry. Meat and Livestock Australian.

MLA, 2013b. Skin Report – Sheep and Lamb. National Livestock reporting service. Meat and Livestock Australia.

MLA, 2013c. Trends in Australian Co-product Values for September Quarter 2013. Meat and Livestock Australia.

Muir, S.K., 2011. Greenhouse Gas Emissions from Australian Beef Feedlots. Department of Agriculture and Food Systems, Melbourne School of Land and Environment. The University of Melbourne, Melbourne, Australia.

National Renewable Energy Laboratory, 2012. U.S. Life Cycle Inventory Database. Accessed 19 November 2012. https://www.lcacommons.gov/nrel/search.

Nguyen, T.L.T., Hermansen, J.E., Mogensen, L., 2010. Environmental consequences of different beef production systems in the EU. J. Clean. Prod. 18, 756–766.

Nguyen, T.T.H., van der Werf, H.M.G., Eugène, M., Veysset, P., Devun, J., Chesneau, G., Doreau, M., 2012. Effects of type of ration and allocation methods on the environmental impacts of beef-production systems. Livest. Sci. 145, 239–251.

Paxton, A., 2011. The Food Miles Report – the Dangers of Long-distance Food Transport. Originally published in 1994 by the Sustainable Agriculture. Food and Environment (SAFE) Alliance.

Pelletier, N., Pirog, R., Rasmussen, R., 2010. Comparative life cycle environmental impacts of three beef production strategies in the upper Midwestern United States. Agric. Syst. 103, 380–389.

Peters, G.M., Rowley, H.V., Wiedemann, S.G., Tucker, R.W., Short, M.D., Schulz, M.S., 2010a. Red meat production in Australia: life cycle assessment and comparison with overseas studies. Environ. Sci. Technol. 44, 1327–1332.

Peters, G.M., Wiedemann, S.G., Rowley, H.V., Tucker, R.W., 2010b. Accounting for water use in Australian red meat production. Int. J. Life Cycle Assess. 15, 311–320.

Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43, 4098–4104.

Pré-Consultants, 2012. SimaPro 7.3 Software (Amersfoort, Netherlands).

Rebitzer, G., Loerincik, Y., Jolliet, O., 2002. Input-output life cycle assessment: from theory to applications 16th discussion forum on life cycle assessment Lausanne. Int. J. Life Cycle Assess. 7, 174–176. April 10, 2002.

Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Glob. Environ. Change 20, 113–120.

Ridoutt, B.G., Sanguansri, P., Freer, M., Harper, G.S., 2012a. Water footprint of livestock: comparison of six geographically defined beef production systems. Int. J. Life Cycle Assess. 17, 165–175.

Ridoutt, B.G., Sanguansri, P., Nolan, M., Marks, N., 2012b. Meat consumption and water scarcity: beware of generalizations. J. Clean. Prod. 28, 127–133.

Rockström, J., Lannerstad, M., Falkenmark, M., 2007. Assessing the water challenge of a new green revolution in developing countries. Proc. Natl. Acad. Sci. 104, 6253–6260.

Stocker, T., Qin, D., Plattner, G.-K., Alexander, S., Allen, N., Bindoff, F.-M., Bréon, J.A., Church, U.C.,S., Emori, P., Forster, P., Friedlingstein, N., Gillett, J., Gregory, D., Hartmann, E., Jansen, B., Kirtman, R., Knutti, K., Kumar, K., Lemke, P., Marotzke, J., Masson-Delmotte, V., Meehl, G., Mokhov, I., Piao, S., Ramaswamy, V., Randall, D., Rhein, M., Rojas, M., Sabine, C., Shindell, D., Talley, L., Vaughan, D., Xie, S.-P., 2013. Technical summary. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: the Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA.

UNEP, 2014. Assessing global land use: balancing consumption with sustainable supply. In: Bringezu, S., Schütz, H., Pengue, W., ÓBrien, M., Garcia, F., Sims, R., Howarth, R., Kauppi, L., Swilling, M., Herrick, J. (Eds.), A Report of the Working Group on Land and Soils of the International Resource Panel. United Nations Environment Programme.

Watts, P., McGahan, E., Bonner, S.L., Wiedemann, S., 2012. Feedlot Mass Balance and Greenhouse Gas Emissions – a Literature Review, Final Report, Project B.FLT. 0361. Meat & Livestock Australia, Sydney, Australia.

Webb, J., Williams, A.G., Hope, E., Evans, D., Moorhouse, E., 2013. Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK? Int. J. Life Cycle Assess. 18, 1325–1343.

Weber, C.L., Matthews, H.S., 2008. Food-miles and the relative climate impacts of food choices in the United States. Environ. Sci. Technol. 42, 3508–3513.

WHO, 2009. Water Sanitation and Health. World Health Organisation.

Wiedemann, S., Henry, B.K., McGahan, E., Grant, T., Murphy, C., Niethe, G., 2015a. Resource use and greenhouse gas intensity of Australian beef production: 1981 to 2010. Agric. Syst. 133, 109–118.

Wiedemann, S., Ledgard, S., Henry, B.K., Yan, M.-J., Mao, N., Russell, S., 2015b. Application of life cycle assessment to sheep production systems: investigating co-production of wool and meat using case studies from major global producers. Int. J. Life Cycle Assess. http://dx.doi.org/10.1007/s11367-015-0849-z.

Wiedemann, S., McGahan, E., Grist, S., Grant, T., 2010a. Environmental Assessment of Two Pork Supply Chains Using Life Cycle Assessment, Project No PRJ-003176 & PRJ-004519. Rural Industries Research and Development Corporation, Barton, ACT.

Wiedemann, S., McGahan, E., Murphy, C., Yan, M.-J., 2015c. Resource use and environmental impacts from beef production in eastern Australia investigated using life cycle assessment. J. Anim. Prod. Sci. 55 (in press).

Wiedemann, S., Yan, M., 2014. Livestock meat processing: inventory data and methods for handling co-production for major livestock species and meat products. In: Paper Submitted to the 9th International Conference of LCA of Food, 8–10 Oct 2014, San Francisco, USA.

Wiedemann, S.G., McGahan, E.J., 2011. Environmental Assessment of an Egg Production Supply Chain Using Life Cycle Assessment, Final Project Report. Australian Egg Corporation Limited, Sydney, Australia.

Wiedemann, S.G., Yan, M.-J., Murphy, C.M., 2015d. Resource use and environmental impacts from Australian export lamb production: a life cycle assessment. J. Anim. Prod. Sci. 55 (in press).

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the Environmental Burdens and Resource Use in the Production of Agricultural and Horticultural Commodities. National Resource Management Institute, Cranfield University and Defra Bedford.