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Resource use and environmental impacts from Australian export lamb production: a life cycle assessment

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Abstract. This study conducted a life cycle assessment (LCA) investigating energy, land occupation, greenhouse gas (GHG) emissions, fresh water consumption and stress-weighted water use from production of export lamb in the major production regions of New South Wales, Victoria and South Australia. The study used data from regional datasets and case study farms, and applied new methods for assessing water use using detailed farm water balances and water stress weighting. Land occupation was assessed with reference to the proportion of arable and non-arable land and allocation of liveweight (LW) and greasy wool was handled using a protein mass method. Fossil fuel energy demand ranged from 2.5 to 7.0 MJ/kg LW, fresh water consumption from 58.1 to 238.9 L/kg LW, stress-weighted water use from 2.9 to 137.8 L H_2O-e/kg LW and crop land occupation from 0.2 to 2.0 m^2/kg LW. Fossil fuel energy demand was dominated by on-farm energy demand, and differed between regions and datasets in response to production intensity and the use of purchased inputs such as fertiliser. Regional fresh water consumption was dominated by irrigation water use and losses from farm water supply, with smaller contributions from livestock drinking water. GHG emissions ranged from 6.1 to 7.3 kg CO_2-e/kg LW and additional removals or emissions from land use (due to cultivation and fertilisation) and direct land-use change (due to deforestation over previous 20 years) were found to be modest, contributing between –1.6 and 0.3 kg CO_2-e/kg LW for different scenarios assessing soil carbon flux. Excluding land use and direct land-use change, enteric CH_4 contributed 83–89% of emissions, suggesting that emissions intensity can be reduced by focussing on flock production efficiency. Resource use and emissions were similar for export lamb production in the major production states of Australia, and GHG emissions were similar to other major global lamb producers. The results show impacts from lamb production on competitive resources to be low, as lamb production systems predominantly utilised non-arable land unsuited to alternative food production systems that rely on crop production, and water from regions with low water stress.

Additional keywords: carbon, footprint, GHG, land, water.

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Introduction

Agricultural systems such as lamb production face the challenge of maintaining and increasing production in the future with constrained natural resources and pressure to reduce environmental impacts from production. 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 (IPCC, Stocker *et al*. 2013). In Australia, water resources are constrained in major river systems such as the Murray–Darling (ABS 2008), and restricted (capped) supply has led to increased competition between water users (MDBA 2012). Arable land resources are limited by soil type and climate to ~4% of national land mass (Lesslie and Mewett 2013). In this context, resource and environmental efficiency is an increasingly important consideration for agri-food systems such as lamb in order to remain competitive in accessing finite resources. Life cycle assessment (LCA) is an internationally recognised tool for determining whole-supply chain resource use and impacts, accounting for multiple impacts concurrently (ISO 2006). Previous Australian sheep LCA have typically been single impact studies (GHG or water) for a single case study farm (CSF) or a small number of farms, producing predominantly Merino wool-meat sheep (Eady *et al*. 2012; Brock *et al*. 2013). Only one study (Ridoutt *et al*. 2012) investigated prime lamb production and this covered water only, using a case study approach. Peters *et al*. (2010*a*, 2010*b*, 2011) performed the only multi-impact study but this covered only one farm in Western Australia (WA) producing lamb for domestic consumption. In other sectors such as beef, recent LCA studies (Wiedemann *et al*. 2015*a*, 2016) have applied a similar multi-impact approach. Our study aimed to address the need for a multi-impact, multi-region analysis of export lamb production to produce a benchmark analysis of resource use and environmental impacts and impact hotspots from cradle to farm gate for Australian export lamb from the three largest production regions. The study used regional datasets to provide a more representative analysis of the study region, and augmented this with detailed case studies to provide specific information regarding on-farm resources and management. The study applied methods for handling co-production based on protein mass allocation, used farm water supply balances to assess fresh water consumption, expanded the assessment of water use to include the impacts of irrigation and applied a disaggregated land occupation assessment based on land type and suitability for cropping. Additionally, the study conducted original research on land use (LU) and direct land-use change (dLUC) emissions from lamb production.

Methodology

Regions

The majority of Australian export lambs are drawn from three major production regions: Victoria (Vic.), South Australia (SA) and New South Wales (NSW). Collectively, northern and southern regions in NSW represent 37% of Australia's sheep flock, Vic. represents 21%, and south-eastern SA represents a further 15% (MLA 2013). In each state, a regional average farm (RAF) was modelled using data from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES 2013) survey of 203 specialist lamb producers, averaged over 5 years (2006–2010). Additional data were obtained from three to five CSF surveyed in each region in the years 2010–2012. The CSF were located in south-western Vic., the southern and northern tablelands of NSW, and from the Hawker region of SA. Four Vic. CSF farms were selected from the Livestock Farm Monitor Project (DPI 2011) based on lamb sale weight (>45 kg) and breeds to suit the export market. NSW and SA CSF data were obtained from three farms surveyed as part of the study, producing export weight first- or second-cross lambs from meat breed or Merino ewe flocks. A simplified description of the production system, showing the system boundary is provided in Fig. 1. Impacts were reported on an intensity basis with a functional unit of 1 kg of liveweight (LW) at the farm gate.

Impacts assessed

The study investigated GHG emissions using the IPCC AR4 global warming potentials of 25 for CH_4 and 298 for N_2O (IPCC 2007). GHG emissions associated with LU and dLUC were included and reported separately, following guidance from the Livestock Environmental Assessment and Performance partnership (LEAP 2014). 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. Fresh water consumption was assessed using methods described and the assessment of stress-weighted water use was based on Pfister *et al*. (2009). Methods for reporting disaggregated land occupation are explained below.

Production and purchased inputs

Production data were accessed from detailed farm records, and verified through discussions with farmers at each CSF. Flocks were modelled over the whole production cycle, from an inventory of breeding ewes, replacement ewes, rams and lambs through to the point of sale. Flocks were modelled with static breeder numbers, with replacement breeder numbers being equal to cull breeder numbers and mortalities combined. Replacement ewes were first mated at 18 months of age across all flocks modelled. Critical factors such as weaning and sale weights were cross checked against 2–3 years of data to remove the effect of unusual seasons. The CSF were predominately pasture-based systems. A proportion of lambs are also finished

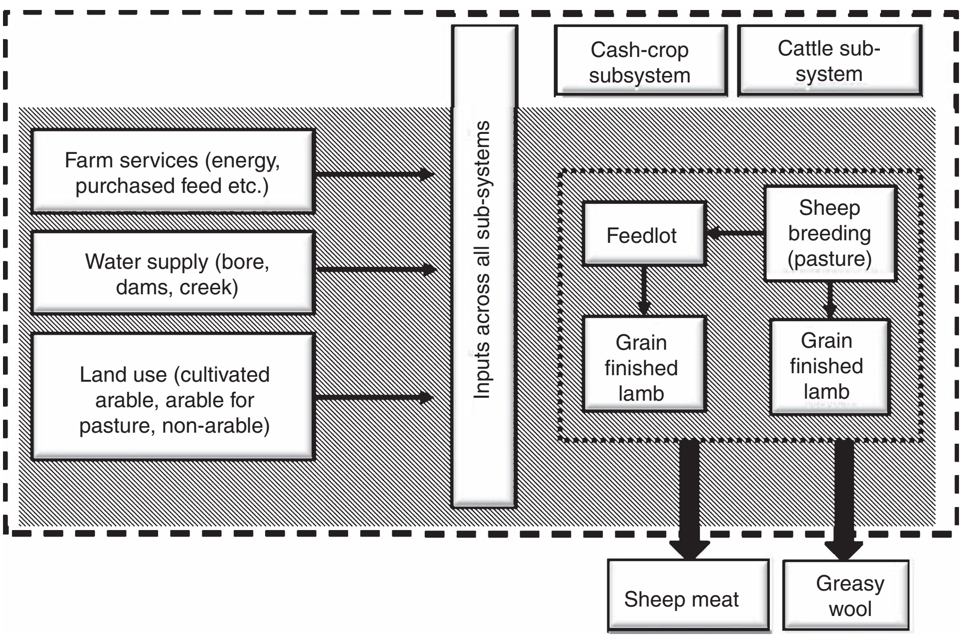


Fig. 1. Generalised outline of the lamb production system showing biophysical inputs and co-products. Shaded area indicates the inputs and processes associated with the sheep flock. Cash cropping and beef cattle subsystems were divided from sheep production and co-production of sheep liveweight and wool was handled using biophysical allocation.

with supplementary grain diets in the southern states, and to account for this, two specialist grain-finishing CSF farms were included in Vic. and SA. Supplementary grain feeding was modelled to represent 15% of all lambs finished in these regions. Flock production data are reported in Table 1. Purchased inputs are reported per tonne of dry matter intake (DMI) by the sheep flock (Table 2) as an aggregate indicator of livestock units. These values can be converted to a dry sheep equivalent basis by dividing by 2.5, assuming a standard 400-kg annual DMI per dry sheep equivalent. Transport of livestock and purchased inputs were included, as was staff travel to and from work. Services (i.e. financial, communications, repairs) were modelled based on expenditure using economic input-output data (Rebitzer *et al*. 2002). Capital infrastructure (i.e. buildings, fences) and machinery were excluded based on their small contribution (<1% of impacts) assessed during the scoping phase. The study required input data at a greater resolution than provided by the ABARES survey for several inputs and productivity factors, requiring additional data and model assumptions. In the absence of ABARES survey data, lamb sale weights were determined from reported lamb sale prices ($/lamb) and market average sale prices ($/kg). Growth rates were determined from lower-bound estimates of lamb age from the corresponding CSF dataset, resulting in growth rates that were intermediate between the CSF and values reported by the DCCEE (2010). Major inputs such as fuel and fertiliser were disaggregated into separate products using the proportions of expenditure from the CSF and market values (available as Supplementary Material for this paper). Impacts generated off-farm via the use of purchased inputs were modelled using background data from the Australian LCI (AustLCI) database where available (Life Cycle Strategies 2007), or the European EcoInvent (2.2) database (Swiss Centre for Life Cycle Inventories 2010). Grains inventory data were modelled using inventory data reported by Wiedemann *et al*. (2010).

Resource use, emissions and removals

Modelling feed intake

Feed intake was modelled using the AFRC (1990) feed intake model as applied by the Australian National Greenhouse Gas Inventory (DCCEE 2010), which determines intake from LW and feed availability. Feed supplement use was determined from records of purchased inputs, and pasture intake was considered to be the residual of predicted feed intake less supplementary feed inputs.

Land occupation

Land occupation was determined using a disaggregated land inventory that accounted for differences in land type using three categories (measured in m^2 year): crop land (arable land used to produce crops), arable land used for pasture (land suitable for crop production currently used for pasture, termed arable pasture land) and non-arable land, which is unsuitable for crop production because of limitations to land capability. For the CSF, the proportion of land in each category was determined

Table 1. Characteristics of lamb production systems for the case study farms (CSF) and regional average farms (RAF) from Victoria (Vic.), New South Wales (NSW) and South Australia (SA)

Parameter	Vic. CSF	Vic. RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Land occupation[A]						
Total utilised land area (ha)	460	453	595	1988	16 061	1096
Crop land (ha)	23	12	33	9	5	12
Arable pasture land (ha)	114	22	20	99	0	54
Non arable land for grazing (ha)	324	419	542	1881	16 056	1030
Farm water supply[B]						
Farm dam (%)	73		64		27	
Bore (%)	19		23		73	
River/creek (%)	8		13		0	
Sheep flock						
Breeding ewes	2364	1309	2220	1255	2733	1171
Breeding ewe culling rate (%)	21.5	23.0	21.1	23.0	29.0	23.0
Breeding ewe mortality rate (%)	4.7	3.1	3.2	4.7	4.0	3.2
Ewe standard reference weight (kg)	67	65	62	65	60	65
Lambing (% at marking)	111.5	98.6	110.5	91.6	82.7	100.8
Average lamb weight at sale (kg LW)	51	50	50	54	52	49
Average daily growth rate of lambs (kg/day)	0.18	0.14	0.16	0.13	0.12	0.14
Annual sheep and lamb sales (kg LW)	134 376	67 312	119 109	66 036	135 598	59 819
Annual wool sales (kg greasy)	18 263	8185	13 557	8972	29 858	7695
LW sold per breeding ewe (kg LW/ewe)	56	51	54	53	50	51
Total flock dry matter intake (t)[A]	1992.3	1084.8	1893.2	1158.6	2406.7	877.9
Biophysical allocation for sheep LW (%)	69	72	73	70	59	71
Economic allocation for sheep LW[C] (%)	81	76	76	69	52	73

[A]Values reported for the total sheep flock and not allocated between wool and LW.
[B]Data for the RAF substituted from the CSF dataset.
[C]Economic allocation was determined from the sale value of sheep and wool, see Supplementary Material.

Table 2. Major purchased inputs of lamb production for the case study farms (CSF) and regional average farms (RAF) from Victoria (Vic.), New South Wales (NSW) and South Australia (SA)
Inputs were expressed per tonne dry matter intake (DMI) for the whole flock

	Vic. CSF	Vic. RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Feed						
Lupins (kg/t DMI)	0.0	5.0	8.0	6.3	0.0	6.5
Hay (kg/t DMI)	12.7	8.0	0.0	10.0	0.0	10.4
Cereal grain (kg/t DMI)	6.1	5.0	36.3	6.3	0.0	6.5
Wheat straw (kg/t DMI)	2.1	3.7	–	–	1.9	4.5
Feedlot ration (kg/t DMI)[A]	8.2	7.2	–	–	5.0	7.4
Energy						
Electricity (kWh/t DMI)	2.2	1.7	2.6	2.2	3.3	2.4
Oil (L/t DMI)	0	0.2	0.0	0.0	0.0	0.2
Diesel (L/t DMI)	4.5	5.2	1.2	4.8	2.4	7.0
Petrol/LPG (L/t DMI)	0.3	0.8	0.9	0.8	0.7	1.0
Fertilisers and soil conditioners						
Superphosphate (kg/t DMI)	19.1	25.0	14.2	18.2	0.0	26.2
Lime (kg/t DMI)	7.8	6.3	21.4	3.1	0.0	1.1
Herbicides (g/t DMI)	142.2	303.2	81.6	97.2	0.0	763.4
Other inputs and services						
Veterinary services ($/t DMI)	9.7	4.3	6.2	5.2	2.8	4.9
Communication services ($/t DMI)	0.7	0.5	0.5	0.7	0.5	0.7
Insurance ($/t DMI)	2.1	1.4	1.3	1.9	1.5	2.0
Accounting ($/t DMI)	1.1	0.8	0.6	1.0	0.7	1.1
Transport distance (km to service centre)	15	30	20	30	110	50

[A]Feedlot ration composition provided in Supplementary Material. Inputs can be converted to whole-flock 'dry sheep equivalent' by dividing by 2.5.

from information provided by the farmers during field observations regarding the suitability of paddocks for cultivation, and land areas were verified with an analysis of satellite imagery. For the RAF, crop land was modelled from the inventory of grain and hay inputs. Total pasture land occupation was reported in the ABARES dataset but land capability was not detailed. In lieu of regionally specific data, arable pasture land in the RAF analysis was determined from an analysis of national statistics from the FAO (FAOSTAT 2014) and Lesslie and Mewett (2013), which showed arable land not currently used for cropping to be 4.9% after Wiedemann *et al*. (2016). No characterisation factors were applied, and land occupation data were reported in square metre years (m^2 year).

Fresh water consumption

The study focussed on fresh water consumption using comprehensive water balance methods recommended by Bayart *et al*. (2010) to assess uses and losses throughout the foreground and background system. 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). Soil stored moisture from rainfall, or so called 'green water' was excluded according to ISO (2014). Water consumption associated with historic LUC was conducted in comparison with a reference system of 1990 (Wiedemann *et al*. 2016) resulting in no change in water yield. This approach provided results that align with the national water accounts (ABS 2012*a*), which consider water consumption based on current LU rather than in comparison to a historic reference system. Water degradation was not assessed because of a lack of regional characterisation and nutrient transport factors for grazing systems across the regions assessed.

Measured water data on-farm were not available and were modelled from an assessment of water supply and utilisation on the farms using water balances, using methods described in Wiedemann *et al*. (2016). Sheep drinking water was estimated using the equation determined by Luke (cited in CSIRO 2007).

$$I_w = 0.1911{*}t - 2.882\ (R^2 = 0.84)$$

Where: I_w = water intake (L/45 kg LW sheep day); t = maximum daily air temperature (°C).

Drinking water estimates per sheep were modified to account for liveweight differences using the method outlined by Luke (1987). This clarification was picked up by the authorship since submission. Drinking water consumed by the animal is lost to the atmosphere via respiration and perspiration, integrated into the product and released outside the river catchment or excreted as urine. Each was modelled as a consumptive use. It could be argued that urination results in a flow of water back to the catchment and is therefore not a consumptive use. However, we consider this water to be analogous to irrigation, because it is deposited to pasture in small, dispersed volumes resulting in a high level of evaporation or transpiration. Proportions of drinking water supplied from bores, creeks and rivers or farm dams (Table 1) were determined from the survey and CSF site visits, and were verified by an analysis of water supply points using satellite imagery. Losses associated with water supply were modelled after Wiedemann *et al*. (2016).

Dam densities were within the range reported by Nathan and Lowe (2012), but extractions for livestock drinking water as a proportion of dam volume were much lower in the present study than the assumptions made by these authors. Farm dam water supply efficiency factors were determined from water balances as an indicator of supply efficiency proportional to total extraction of water from the environment. These factors were 17.5% (Vic. CSF), 18.2% (NSW CSF) and 7.5% (SA CSF). In the absence of farm water data in the ABARES dataset, supply data and dam supply efficiency from the CSF datasets were applied.

Irrigation water use for lamb production was determined from irrigation land reported by the ABARES survey and irrigation rates from the ABS survey of water use on Australian farms (ABS 2012*b*). None of the CSF used irrigation. Total estimated irrigation water use was verified by comparison with reported irrigation for 'sheep and other livestock' for the years 2007/8–2009/10 (ABS 2011). Water use per kg LW from both datasets was in agreement, providing confidence in the estimates based on the ABARES dataset. Irrigation associated with feed grain inputs was modelled using the water inventory of Ridoutt and Poulton (2009) and total irrigation of cereal crops in each state reported by the ABS (2012*b*). Losses associated with irrigation water supply of 27.1% were applied, based on the ABS national water accounts (ABS 2012*b*).

Stress-weighted water use

The stress-weighted water use was assessed following Pfister *et al*. (2009) by accounting for the expected impact of water use in a given catchment using a global stress weighting factor. To calculate the stress-weighted water use, fresh water consumption was multiplied by the regional water stress index (WSI, Pfister *et al*. 2009) and was then divided by the global average WSI (0.602) and expressed as water equivalents (L H_2O-e.) following Ridoutt and Pfister (2010).

Fossil fuel energy demand

Fossil fuel energy demand was associated with both direct energy demand on-farm and energy demand from the manufacture and transport of goods and services used by the farms. Modelling of energy demand was based on the inventory of purchased goods, services (Table 2) and transport distances, using inventory data from AustLCI or EcoInvent (Life Cycle Strategies 2007; Swiss Centre for Life Cycle Inventories 2010).

Greenhouse gas emissions

Emissions from fossil fuel energy were modelled directly from the energy inventory. Livestock GHG emissions (enteric and manure) were modelled using Australian National Greenhouse Gas Inventory methods (DCCEE 2010) (Table 3).

Greenhouse gas emissions from LU and dLUC

Soil carbon (C) changes under crop and pasture soils (LU emissions and removals), were included following guidance from LEAP (2014). Soil C losses from crop land were mainly related to the use of purchased grain sourced regionally. Estimated soil C losses took into account the different rate of loss from conventional and zero-tillage (Dalal and Chan 2001; Chan *et al*. 2003), assuming multiple cultivations occur on 37% of crop land (NSW, Vic., SA – ABS 2009) with remaining land being zero-tillage. Soil C losses were assumed to be 0.1 t C/ha.year for zero tillage and 0.58 t C/ha.year 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 cropping soils, soil C sequestration rates of 0.29 ± 0.17 t C/ha.year (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 C 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 and Koen 2013; Schwenke *et al*. 2013). Phosphorus fertiliser application was common in the sheep systems studied (Table 2) and C removals were explored using two scenarios: (i) zero change in soil C under fertilised pasture, (ii) or C sequestration of 7.2 t C/ha under fertilised pasture, based on the sequestration rate of 0.29 t C/ha.year in Sanderman *et al*. (2010) over a 25-year period corresponding to establishment of a new equilibrium soil C level. Carbon removals from sequestration were annualised over a 100-year period to align with the requirements for permanence. 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 3-year fertiliser rotation.

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. Although deforestation in southern states of Australia has fallen to very low levels since 1990 (DCCEE 2012), historic emissions must be considered. Conversion of forest to grassland

Table 3. Greenhouse gas prediction methods used in the study

Emission source	Key parameters/model	Reference
Enteric CH_4 (kg/sheep)	kg dry matter/head × 0.0188 + 0.00158	DCCEE (2010)
Manure CH_4 (kg CH_4/kg dry matter manure)	1.4×10^{-5}	DCCEE (2010)
Manure N_2O (kg N_2O-N/kg N)		DCCEE (2010)
In urine	0.004	
In faeces	0.005	
Manure NH_3 (kg NH_3-N/kg N excreted in manure)	0.2	DCCEE (2010)
Indirect N_2O from NH_3 losses (kg N_2O-N/kg NH_3-N volatilised)	0.004	Derived from DCCEE (2010)
Indirect N_2O from leaching and runoff (kg N_2O-N/kg NH_3-N volatilised)	0.0075	De Klein *et al*. (2006)

attributable to sheep was assumed to be negligible, because of the dramatic decline in Australian sheep numbers from 170 million in 1990 to 68 million in 2010 (ABS 2013), 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 12%, 21% and 20% for NSW, Vic. and SA, respectively, based on comparison of the maximum reported area of land cultivated for cereal crops in the 5 years before 1990, with the maximum reported area cultivated in the 5 years to 2010 (ABS 2013). Direct LUC was predominantly from conversion of grassland (DCCEE 2012), assumed to be 80% (NSW) and 95% (Vic., SA) of new crop land, with the remainder arising from conversion of forest land. Total C 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 2012), corresponding to annualised emission rates of 15.5 and 2.3 t CO_2-e/ha.year. Divided over the total crop land, dLUC emissions per hectare were 0.61, 0.61 and 0.58 t CO_2-e/ha.year for NSW, Vic. and SA, respectively.

Handling co-production

Following the recommendation of ISO 14044 (ISO 2006), inputs to the farming systems were first divided among farm subsystems (sheep, beef, cropping) and were accounted separately. Inputs that were not specific to a particular subsystem, such as administration overheads or fertiliser inputs to pasture consumed by both sheep and cattle, were divided based on proportion of annual feed intake by each species. Cattle DMI was predicted from herd data in the CSF and ABARES datasets and feed intake prediction methods from the NGGI (DCCEE 2010). System separation methods for the ABARES dataset are described in the Supplementary Material. Within the sheep subproduction system, LW from lambs and culled breeding animals was not differentiated, because there are no significant biophysical or nutritional differences between products from these different animals. To handle allocation of greasy wool and total flock LW, we applied a biophysical method based on protein mass allocation (Wiedemann *et al*. 2015*b*). Protein mass was determined from clean wool yield (68%), and an assumed 100% protein content (DM basis) for clean wool. Protein content in LW (18%) was determined using the regression equation of Sanson *et al*. (1993). Flock allocation factors are provided in Table 1. To check the sensitivity of the allocation method, economic allocation and system expansion were also applied for comparison. LW and greasy wool sale values were reported by the farmers, and were averaged over multiple seasons to reduce variation, with average allocation values reported in Table 1. The system expansion method was applied using the avoided burden method, using nylon fibre as the avoided product to replace wool (see Wiedemann *et al*. 2015*b*). All impacts were attributed to LW, and impacts associated with the manufacture of an equivalent mass of nylon were subtracted from the impacts from sheep production. Inventory values for nylon were taken from EcoInvent (Swiss Centre for Life Cycle Inventories 2010).

Results

Resource use

Crop land occupation ranged from 0.2 (SA CSF) to 2.0 (NSW CSF) m^2 year/kg LW (Fig. 2). Crop land was up to 6% of total land occupation with the majority of land being classified as non-arable. Non-arable land occupation varied from 16.7 (Vic. CSF) to 697.4 m^2/kg LW (SA CSF) with the intensity of non-arable land occupation corresponding to relative rainfall and pasture production. The semiarid SA CSF region (697.4 m^2 year/kg LW) was characterised by large areas of native pasture and very low stocking rates, which is reflected in the unusually high land occupation value.

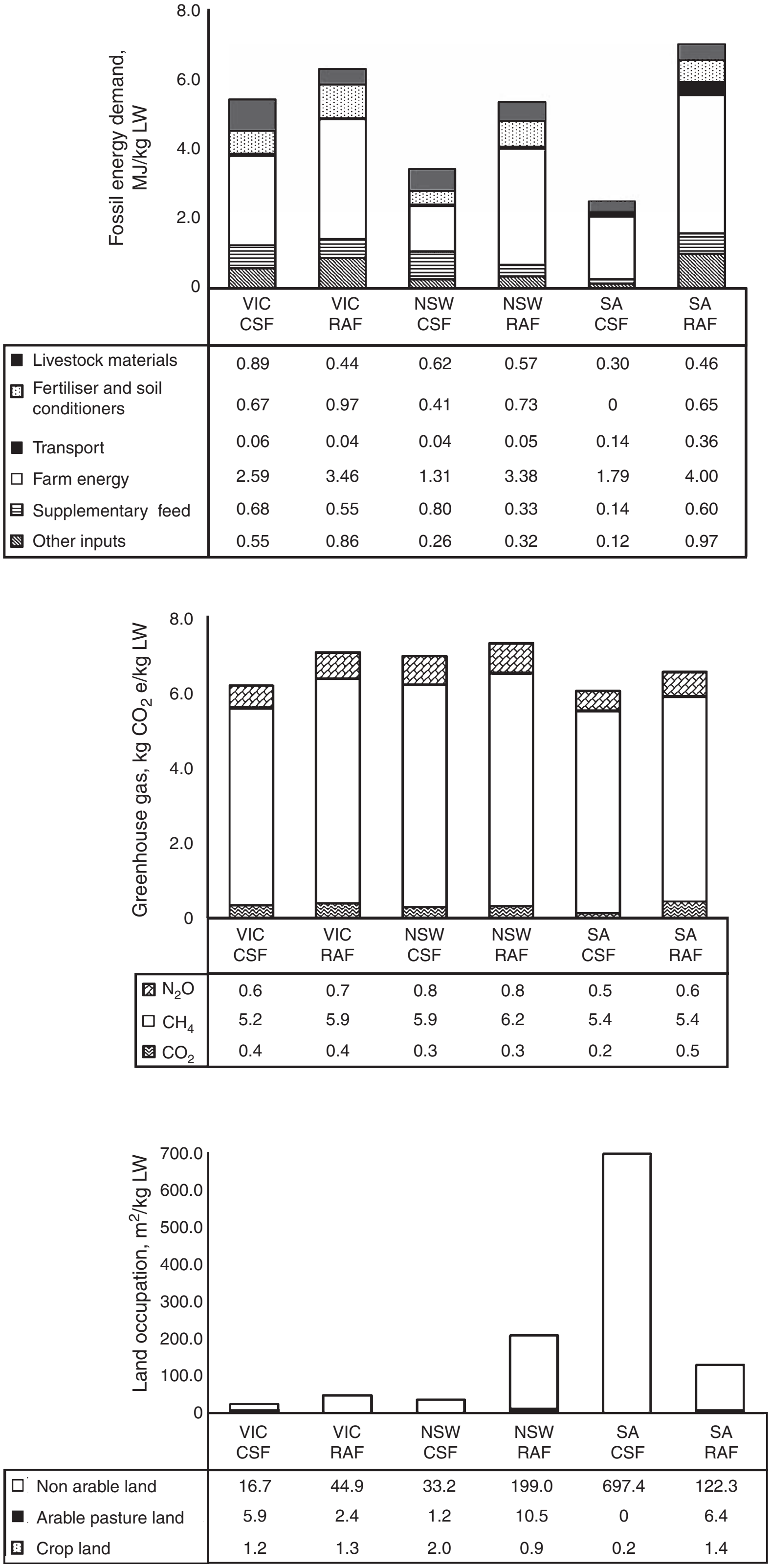
Fresh water consumption (Table 4) ranged from 58.1 to 238.9 L/kg LW and non-weighted mean values from the CSF across all regions (71.7 L) were one-third of the non-weighted mean RAF results (218.5 L/kg LW). The primary differences between the datasets related to the utilisation of irrigation water and associated supply losses, which did not occur on any CSF but contributed 44%, 72% and 73% for the SA, NSW and Vic. RAF analyses, respectively.

Losses associated with the supply of drinking water ranged from 39.4 to 100.4 L/kg LW. Variability in supply efficiency was influenced by water supply source and net evaporation rates. Higher reliance on dams resulted in higher loss rates compared with bore or creek water supply, and dam supply losses varied with climatic factors such as net evaporation. Livestock drinking water ranged from 12.9 to 29.0 L/kg LW, with differences related primarily to temperature and to a lesser extent productivity.

Total fossil fuel energy demand ranged from 2.5 (SA CSF) to 7.0 (SA RAF) MJ/kg LW (Fig. 2). There was a trend towards higher energy demand for the RAF, which corresponded to higher levels of purchased farm inputs (SA) and on-farm fuel use (NSW) than the corresponding CSF. Farm energy demand from consumption of diesel, petrol and electricity contributed 38–72% of total energy demand (Fig. 2) and was the largest input. When averaged across datasets, the energy demand associated with fertiliser, livestock materials and supplementary feed were very similar, each contributing 10–11% to total energy demand. In contrast, transport was a minor contributor, averaging only 2.3% of total energy demand.

Impacts

Stress-weighted water use differed considerably between regions and datasets, ranging from 2.9 (SA CSF) to 137.8 (NSW RAF) L H_2O-e/kg LW (Table 4). The wide range was mainly the result of large differences in water stress, from very low (parts of Vic., 0.0107) to high (parts of NSW, 0.815). Weighted average values were intermediate between these levels for the RAF. 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. However, the WSI is a global index with coarse local resolution, and caution should be applied in drawing conclusions at the state level. GHG emissions (excluding LU and dLUC) ranged from 6.1 (SA CSF) to 7.3 (NSW RAF) kg/kg LW (Fig. 2). The GHG emissions profile was dominated by enteric CH_4 from the sheep flock (83–89%) followed by N_2O and smaller emissions of CO_2 from fossil fuel energy demand and lime application. The two scenarios for LU and dLUC resulted in either a small emission source (ranging from zero to 0.3 kg CO_2 eq/kg LW), where soil C removals were assumed to be zero, or modest removals of –0.9, –1.0 and –1.6 (NSW, Vic. and SA RAF), and to between –0.3 (NSW and



	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Livestock materials	0.89	0.44	0.62	0.57	0.30	0.46
Fertiliser and soil conditioners	0.67	0.97	0.41	0.73	0	0.65
Transport	0.06	0.04	0.04	0.05	0.14	0.36
Farm energy	2.59	3.46	1.31	3.38	1.79	4.00
Supplementary feed	0.68	0.55	0.80	0.33	0.14	0.60
Other inputs	0.55	0.86	0.26	0.32	0.12	0.97

	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
N_2O	0.6	0.7	0.8	0.8	0.5	0.6
CH_4	5.2	5.9	5.9	6.2	5.4	5.4
CO_2	0.4	0.4	0.3	0.3	0.2	0.5

	VIC CSF	VIC RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Non arable land	16.7	44.9	33.2	199.0	697.4	122.3
Arable pasture land	5.9	2.4	1.2	10.5	0	6.4
Crop land	1.2	1.3	2.0	0.9	0.2	1.4

Fig. 2. Fossil fuel energy demand (direct and indirect), greenhouse gas emissions (excluding land use and land-use change) and land occupation per kilogram of liveweight at the farm gate for lamb produced from the case study farms and regional average farms from Victoria, South Australia and New South Wales.

Table 4. Contribution to fresh water consumption and stress-weighted water use per kilogram of liveweight (LW) at the farm gate for lamb produced from the case study farms (CSF) and regional average farms (RAF) from Victoria (Vic.), New South Wales (NSW) and South Australia (SA)

Water use	Vic. CSF	Vic. RAF	NSW CSF	NSW RAF	SA CSF	SA RAF
Livestock drinking water (L/kg LW)	12.9	13.2	16.3	13.5	22.0	29.0
Drinking water supply losses (L/kg LW)	42.7	44.3	41.2	39.4	76.2	100.4
Irrigation (L/kg LW)	0.0	127.0	0.0	113.4	0.0	84.3
Irrigation supply losses (L/kg LW)	0.0	33.0	0.0	29.5	0.0	21.9
Other minor uses (L/kg LW)	2.5	2.1	1.0	1.4	0.1	3.2
Total fresh water consumption (L/kg LW)	58.1	219.6	58.6	197.1	98.3	238.9
Stress-weighted water (L H_2O-e/kg LW)	5.9	86.9	20.7	137.8	2.9	8.8

Vic. CSF) and 0.0 kg CO_2 eq/kg LW (SA CSF) when the soil C sequestration under fertilised pastures was included in the model.

Sensitivity of model assumptions

Methods applied to handle the co-production of wool and LW can result in large differences in impacts for each product in sheep systems (Wiedemann *et al*. 2015*b*). To improve the transparency of the analysis, results are also presented for greasy wool in the Supplementary Material, and sensitivity of allocation method choices were explored using two comparative methods. Economic allocation resulted in 4–14% higher impacts attributed to LW, and associated lower impacts for greasy wool across most cases. In contrast, using system expansion, increased GHG, fresh water consumption, stress-weighted water and land occupation by 37–48%, but reduced fossil fuel energy demand by 254%. This variation between impact categories when applying system expansion relates to the high energy intensity of the avoided nylon product but relatively lower impacts across other categories.

To explore the sensitivity of enteric CH_4 assumptions, we remodelled results applying the IPCC 2006 tier II method (Dong *et al*. 2006), which resulted in a ~4% reduction in GHG in response to the lower emission factor for lambs applied by the IPCC. Emissions or removals from soil C are less well understood than other emission sources for sheep production, with no published sheep LCA addressing these losses to the author's knowledge. Higher dLUC emissions may exist from crop land because of variation in soil C loss, though this will only have a small impact on lamb because of the small amount of crop land required for lamb production. Alternative assumptions for soil C change under pasture may have a greater effect. If higher levels of soil C sequestration such as the 9.9 t C/ha reported by Chan *et al*. (2010) could be achieved, removals could be ~35% higher than reported in the 'high' sequestration scenario. However, this is unlikely to be observed in lower rainfall and mixed cropping regions (Davy and Koen 2013).

Discussion

Resource use and impact intensity

Investigating resource use and impacts relative to production using an intensity metric presents an effective way to balance the needs for maintained or increased production with the pressure of limited and elevated environmental impacts. Weighted mean results from the regional analyses (using regional sheep flock sizes of Vic., NSW and SA) were 7.1 kg CO_2-e, 6.0 MJ, 1.1 m^2, 212.2 L and 96.4 L H_2O-e/kg LW for GHG (excluding LU and dLUC), fossil fuel energy, crop land, fresh water consumption and stress-weighted water use, respectively. GHG emissions intensity was higher than previous Australian studies that focussed on Merino sheep systems rather than prime lambs, that showed GHG emissions of 5.3 and 6.2 kg CO_2-e/kg LW (Eady *et al*. 2012; Brock *et al*. 2013). Peters *et al*. (2010*a*) reported impacts of 4.4–4.7 kg CO_2-e/kg LW for Merino cross-bred lambs in WA. These low values correspond to higher allocation of impacts to greasy wool as each of these studies applied economic allocation to lambs produced from Merino sheep where greasy wool value is high. Although different allocation methods alter results between the two products, this does not alter the overall efficiency of the system; it simply shifts the impacts between the two products. In the studies cited, impacts from wool are higher than reported here indicating the impacts were transferred to the wool product. Improved flock productivity will reduce GHG emissions intensity from animal sources (Alcock and Hegarty 2011), which dominate the emissions profile of lamb. This is achieved by improving flock feed efficiency via the 'dilution of maintenance' effect (Johnson *et al*. 1996). In the present study, emission intensities were lowest from flocks with the highest lamb production per breeding ewe (Vic. CSF) or high lamb and greasy wool production per breeding ewe (SA CSF), suggesting that a focus on either lamb production or dual-purpose lamb-wool can result in low emissions intensity. Inclusion of LU and dLUC emission and removal sources showed these to be a minor source of emissions, adding up to 4% to total GHG in the scenario where no soil C sequestration was assumed to occur under pasture. Where sequestration was assumed to occur, mean emissions in the RAF were 16% lower than when LU and dLUC were excluded. These impacts have not been assessed in previous studies, and the present results suggest that soil C sequestration under fertilised pastures could be a significant removal if achieved, though as noted this is challenging in the Australian context and the results have a high degree of uncertainty.

Fossil fuel energy demand was found to be 37% and 54% lower in the NSW and SA CSF compared with Vic. CSF and differences were observed between farms in the CSF dataset. Energy demand is a function of production intensity and was most strongly related to on-farm fuel used for vehicles and machinery. We found fertiliser manufacture to range from 9% to 15% of energy demand with the exception of the extensive SA CSF where no fertiliser was used. Livestock materials, which were modelled as veterinary inputs using input-output data, were found to be a significant contributor to energy demand, ranging from 7% to

18%. Production of supplementary feed was also a significant though variable input, contributing from 6% to 23% of on-farm energy demand. Farms that used fodder or grain required higher energy inputs.

The variation in energy demand between individual farms suggested that opportunities may exist to reduce direct energy demand on sheep farms. Interestingly, we found that farms with lower stocking densities corresponded to lower inputs of energy, and did not result in large differences in GHG emissions. This was partly in response to compensatory factors; for example lower energy demand resulted in reduced GHG. However, lower GHG impacts in the SA CSF analysis were also partly related to high wool production and correspondingly high allocation of impacts to wool. Modelled fresh water consumption included assessment of sources not previously considered in Australian studies, including supply losses from farm dams, and irrigation associated with pastures and purchased inputs. Supply losses and irrigation water use dominated fresh water consumption, rather than drinking water. With irrigation removed, water use was similar to the values reported by Peters *et al*. (2010*b*) of 64–100 L (converted to LW basis) for lambs produced in WA though this excluded losses associated with water supply and irrigation. Ridoutt *et al*. (2012) reported water use of 39 L and stress-weighted water use of 14.7 L H_2O-e/kg LW for lamb production in Vic. but excluded irrigation, and assumed much lower supply losses from farm dams than our study. The present results highlight the significance of irrigation and supply losses and suggest water use is higher than previously reported. Drinking water is essential for production and cannot be reduced easily. Supply losses from farm dams can be reduced using improved infrastructure (larger and deeper dams) but these strategies are unlikely to be cost effective. Considering the large volume of irrigation water use, this represents a potential target for reducing water use across the industry though reduced feed production from irrigation land must be offset by increases from dryland areas to maintain total production. Trade-offs may also exist between resource use and environmental impacts. Increasing production from dryland areas may require increased inputs from fertiliser to increase pasture production, or utilisation of grain. Productivity improvements can be achieved via genetic improvement and improved animal husbandry, but high quality feed is also a requirement for high reproductive efficiency and growth rates. Methods to produce higher quality feed are often energy intensive (i.e. fertiliser inputs, fuel inputs for cropping) and may utilise different resources such as crop land and irrigation, and hence, comprehensive analyses are required to understand trade-offs. Such trade-offs also exist when considering lamb in the context of other food production systems. For example, some researchers (i.e. Garnett 2011) have promoted replacing red meat with vegetable protein to lower environmental impacts, but the lamb production systems studied here relied predominantly on non-arable land resources, which is unsuitable for producing vegetable protein or feed grain needed as inputs for monogastric meat production systems.

Australian lamb in the global context

The regional analysis showed similar impacts between states for GHG and water, whereas energy and stress-weighted water use varied to a greater extent. Webb *et al*. (2013) reported emissions of 7.3 kg CO_2-e/kg LW (converted from carcass weight) for lamb produced in the UK whereas Ledgard *et al*. (2011) estimated GHG emissions of 8.6 kg CO_2-e/kg LW for NZ lamb. Both results were similar or slightly higher than the average results presented here, and differences in allocation method may explain some variation. Although our study only investigated impacts to the farm gate, Wiedemann *et al*. (2015*c*) showed that pre-farm-gate impacts of Australian lamb represent ~89% of impacts through to retail in the USA, which is similar to findings for New Zealand export lamb (Ledgard *et al*. 2011). Webb *et al*. (2013) demonstrated that lamb imported from NZ may have lower impacts than lamb produced and sold in the UK market, highlighting that differences at the production level are more significant than export distance. None of the studies reviewed included potential GHG emissions from LU and dLUC. Our analysis suggested that these sources may be a small emission or removal depending on assumptions regarding soil C change under pasture.

Fewer studies have investigated land occupation or fossil energy demand. Total land occupation of non-organic lamb produced in the UK was found to be 6.5 m^2/kg LW (converted from carcass weight – Williams *et al*. 2006). This is significantly lower than the total land occupation of the present study in response to the higher pasture productivity of UK grazing land. Although land occupation was much higher in our study, the majority of land occupation was from non-arable land. In Australia, non-arable land is the largest land resource in the country (Lesslie and Mewett 2013) and from an agricultural perspective is suitable only to ruminant grazing systems.

Reported energy use for lamb production in the UK ranged from 9.7 to 12 MJ/kg LW (converted from carcass weight; Williams *et al*. 2006; Webb *et al*. 2013), which tended to be higher than results presented here, particularly in comparison to the CSF dataset. However, these studies included energy demand from renewable sources which were excluded from the present study, which may explain why energy demand was slightly higher in these studies.

Larger differences in water use were observed with studies applying different assessment methods. For example, Mekonnen and Hoekstra (2012) reported virtual water content (VWC) values ranging from 2790 to 5427 L/kg LW lamb (converted to LW basis). Of the total virtual water content, over 90% was green water; soil stored moisture from rainfall in dryland systems. So called ‘blue water’, which is broadly comparable to fresh water consumption in the present study, was 151–279 L/kg LW with a global average of 250 L/kg LW. Supply system losses were not included in these VWC estimates (M. M. Mekonnen, pers. comm.) and these may add additional fresh water consumption to this global analysis depending on the water supply systems used.

Conclusions

This study presents the first multi-criteria resource and environmental impact analysis of farm-gate export lamb production from the three major production regions in eastern Australia. The study applied new methods for handling co-production, estimating water use and GHG from LU and dLUC, and provided disaggregated land occupation results not previously reported. The analysis of fresh water consumption water use found supply losses and irrigation to be significant losses not previously analysed in detail by lamb LCA. Despite inclusion of additional uses, fresh water consumption was lower than less comprehensive global analyses of blue water use for lamb production. Lamb production used moderate-low amounts of fossil fuel energy and crop land per kilogram of product. Lamb production relied predominantly on non-arable land not suitable for many alternative food production systems that rely directly or indirectly on arable land. GHG emissions were dominated by animal-related emissions, with smaller contributions from CO_2 or indirect emission sources. This suggests that improving flock productivity will be most effective in reducing emissions intensity. Soil management was also identified as a potential source of emissions or removals depending on LU, and further research is required to develop a robust understanding of these factors.

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