Red Meat Production in Australia: Life Cycle Assessment and Comparison with Overseas Studies

GREGORY M. PETERS,[†]
HAZEL V. ROWLEY,[†]
STEPHEN WIEDEMANN,[§]
ROBYN TUCKER,[§] MICHAEL D. SHORT,[†]
AND MATTHIAS SCHULZ*[,†]

UNSW Water Research Centre, The University of New South Wales, UNSW Sydney, NSW 2052, and FSA Consulting, Toowoomba, Queensland, Australia

Received July 13, 2009. Revised manuscript received November 16, 2009. Accepted December 10, 2009.

Greenhouse gas emissions from beef production are a significant part of Australia's total contribution to climate change. For the first time an environmental life cycle assessment (LCA) hybridizing detailed on-site process modeling and input–output analysis is used to describe Australian red meat production. In this paper we report the carbon footprint and total energy consumption of three supply chains in three different regions in Australia over two years. The greenhouse gas (GHG) emissions and energy use data are compared to those from international studies on red meat production, and the Australian results are either average or below average. The increasing proportion of lot-fed beef in Australia is favorable, since this production system generates lower total GHG emissions than grass-fed production; the additional effort in producing and transporting feeds is effectively offset by the increased efficiency of meat production in feedlots. In addition to these two common LCA indicators, in this paper we also quantify solid waste generation and a soil erosion indicator on a common basis.

Introduction

On the basis of high-level estimates, greenhouse gas (GHG) emissions from agriculture constituted 16% of Australia's total GHG emissions in 2006 (*1*). Australia's livestock industries are considered responsible for about 70% of the agriculture sector's emissions and 11% of the national total emissions. These data highlight both the significant contribution of livestock industries to Australia's GHG emissions profile and the correspondingly significant opportunity to enhance the environmental performance of the Australian economy through industry-targeted measures. To achieve optimal environmental outcomes and target management interventions, managers and policy-makers need performance information that takes a holistic "life cycle" perspective and is based on best practice data acquisition and analysis.

Life cycle assessment (LCA) is a form of cradle-to-grave systems analysis that attempts to quantify the important environmental impacts of all processes involved in a production system. The importance of quantitative environmental data and LCA for meat products is recognized worldwide, with LCAs having been conducted on pork (*2, 3*), lamb (*4, 5*), and beef (*4–15*). Related LCAs on leather (*16*) and particularly milk and dairy products (*6, 8, 13, 17–23*) have also been the subject of intense investigation during the past decade. Northern European beef is primarily sourced from dairy cattle, and many studies grapple with the complexity of allocating environmental impacts to milk, meat, and other coproducts such as leather. Feed choice (and related enteric methane emissions) and feed production systems are the major contributors to the environmental impact in all cases.

When examining modern red meat production, including the grain used in the lot-feeding sector is essential. While most grain is produced from dryland (no irrigation) crops in Australia, pesticides and herbicides used during production and storage must be considered in LCA (*24*). Oilseeds and protein meals are important feedstuffs in the lot-feeding sector and are also commonly used in drought feeding situations on grazing properties. Detailed multiorder process life cycle inventory (LCI) data for wheat and canola production have been compiled for Australian cropping systems (*25*).

Wood et al. (*26*) primarily used first-order process data and an input–output analysis (IOA) approach for higher orders to examine organic and conventional farming practices in Australia in a hybrid input–output LCA. Their examination of meat, grain, fruit, and vegetable production showed that while the direct on-site use of energy and materials of organic farms is higher than at the conventional farms, organic farming performs better when the entire supply chain is considered, except in the case of sheep and wheat production. Their calculations, however, excluded enteric methane—the principal system GHG.

A recent Swiss study (*7*) compared pasture and feedlot beef production systems and concluded that the differences between the two systems are relatively minor, except for the considerably greater ecotoxicity impacts for the feedlot system resulting from greater fertilizer use. Various other studies have been published to enhance the LCI analysis of water use (*27–30*) as well as energy use and GHG emissions (*31–35*) on cattle grazing properties and feedlots.

The aim of this work was to examine red meat production to improve the accuracy of some previous research in two ways. First, we used detailed process analysis focusing on three particular supply chains in two operating years (2002 and 2004), including the use of measured animal growth rates, rather than relying on regional averages as per the standard method (*32*). Second, we removed the system boundary problem afflicting some studies by applying a tiered hybrid input–output LCA. We examined the systems for primary energy consumption, GHG emissions ("carbon footprint"), solid waste production, soil erosion potential, nutrient balance, soil acidification potential, eutrophication potential, and water use. In this paper we focus on the first four of these indicators.

Materials and Methods

System Description. The red meat production systems under consideration are (1) a sheepmeat supply chain in Western Australia (WA), (2) a beef supply chain in Victoria (VIC) producing organic beef, and (3) a premium export beef supply chain in New South Wales (NSW). The majority of cattle from the NSW supply chain spend 110–120 days at a feedlot before being sent to the meat processing plant. Data were collected for the 2002 and 2004 calendar years for each supply chain.

* Corresponding author e-mail: m.schulz@unsw.edu.au; phone: +61 2 9385 5945; fax: +61 2 9313 8624.

[†] UNSW Water Research Centre, The University of New South Wales.

[§] FSA Consulting.

10.1021/es901131e
Published on Web 01/12/2010

The basis for this study, called the "functional unit" (FU) (*36*), is "the delivery of 1 kg of hot standard carcass weight (HSCW) at the exit gate of the meat processing plant". The system boundary encompasses all on-site and upstream processes at the farm, feedlot, and whole processing plant, including transport between these sites. Environmental emissions associated with transport after the meat processing plant and other processing, retailing, or consumption activities are outside the system boundary of this study.

Inventory Data. LCI data for the three grazing properties and the NSW feedlot operation include all recurrent energy use (electricity and fuels), material inputs (e.g., feeds, fertilizers, pesticides, soil modifiers), water use, and solid waste production. To quantify each grazing property's production in terms of the FU, data on stock purchases, sales, transfers, births, and deaths and liveweights, growth rates, and dressing percentages were collected. Quantitative LCI data for a typical meat processing plant were obtained from published industry data (*37*). For all three supply chains, the regional differences in the environmental profile of electricity production were considered. For fuel and transport processes, Australian data sets were used. Data for wheat, barley, canola (*25*), hay, and mineral supplement (*38*) production were taken from previous LCI work. In the absence of specific LCI data for triticale, lupins, and unspecified grain, production was modeled as being identical to that of wheat. For remaining feed materials such as fluffy cottonseed and tallow and for some pesticides and other chemical inputs for which LCI data were not readily available, an IOA approach was applied. Regarding molasses, fertilizers, and soil modifiers, Australian data sets—or if unavailable international data sets—were used and modeled in the LCA software GaBi, version 4.2 (*39*).

To assess the environmental impacts of the extended supply chain, which is difficult using a process-based LCA approach, IOA was used. IOA involves constructing a mathematical model of the national economy and the environmental impacts of industries (*40*). In the first step, the expenditure data are assigned to industry sectors defined by the Australian Bureau of Statistics (*41*). Second, to avoid the potential for double-counting, the location of the interface between the process-based and IOA parts of the model is defined (*40*). Finally, the environmental interventions (e.g., extractions and emissions) are linked to the industry sectors under consideration (*40*) and in accordance with the respective expenditure data.

Life Cycle Impact Assessment. GHG emissions were estimated including all aspects of red meat production such as on-farm energy consumption, enteric processes, manure management, livestock transport, commodity delivery, water supply, and administration. Regarding GHG emissions from livestock on grazing properties and feedlots, calculations were based on the methodology described by the National Greenhouse Gas Inventory Committee (NGGIC) (*42*) enhanced by primary farm animal growth rate and mass data. The relative contribution of each GHG to the carbon footprint were estimated by using equivalence factors set in the most recent publication by the Australian Government (*43*), which uses factors agreed upon for the Kyoto Protocol.

Energy consumption was estimated from data supplied by individual producers. Factors provided by the Australian Greenhouse Office were used to convert raw energy data (e.g., liters of diesel) into primary energy consumption (e.g., megajoules of primary energy) (*34*). Primary energy is also referred to as "full cycle" energy and means, for example, that electricity consumption is related back to both the coal burned to generate it and the energy involved in obtaining the coal.

Solid wastes generated on livestock grazing properties include tires, chemical containers and drums, end-of-life vehicles and equipment, and organic waste (e.g., mortalities, spoiled feed). Solid wastes are often disposed of in on-farm tips. Waste production by goods and services providers is also included in the overall analysis. Feedlot manure is not considered a waste since it is spread on paddocks as a fertilizer substitute and does not leave the LCA system boundary. The data set for the meat processing plant excludes the paunch and yard manure as wastes since these are recycled for soil improvement additives.

Soil erosion is a natural resource management issue of particular concern to the Australian red meat industry. Soil erosion by water was assessed using broad-scale erosion mapping (*44*), which takes into account the erosion gully density and the annual hillslope erosion rate. This mapping was ground-truthed by a qualitative assessment of the topography and management system used on each property. To avoid attributing natural erosion processes to red meat production, erosion rates were compared to natural or pre-European erosion rates. The estimates presented in this paper indicate the erosion risk on land used for red meat production in different regions of Australia. We estimate this on the basis of a second FU: "1 ha of land used for production", to avoid attributing the soil erosion impact solely to red meat production when it is related to many extraneous factors. For further details on soil erosion data sources and assumptions, refer to the Supporting Information.

Quantifying Production. For the lot-feeding property, net HSCW gain is readily modeled as the difference between outgoing and incoming HSCW; however, in the context of a grazing property, the "production cycle" exceeds the annual study period and stocking rates can vary widely. For example, a grazier may choose to hold off selling or buying stock due to market price fluctuations or environmental factors such as feed availability. Hence, there was a need to consider production of carcass weight not sold during the study years. To that end, growth rates for different cohorts and classes of livestock were identified. All livestock were distributed into these categories, and the final weight gain from the property data was determined using a livestock production model that considers the growth rate of the animals and the number of days the animal spends in that group on the farm or feedlot. The mass of livestock imported to the properties is excluded from the calculation of the system's productivity since the growth of purchased stock occurs in various locations for which detailed, energy, GHG, or soil erosion data are unavailable or unavailable in the study years and for which individual property production data would be needed to partition emissions such as enteric and petrochemical GHGs. Consequently, the red meat produced outside the defined system and the environmental burdens associated with this production are consistently excluded from the model.

Allocation. Whenever systems under study generate more than one saleable output, allocation is needed to assign the environmental impacts to the FU. Mass-based and economic allocations were applied at the scale of individual process units (e.g., grazing property, feedlot, processing) because relative production rates vary between the process units. The grazing properties under study produced HSCW and non-HSCW liveweight products as well as other products such as wool and grain. Data collected from each property on the relative mass and economic value of outputs produced were used to assign an appropriate proportion of the LCI to the HSCW production system. Year-adjusted commodity prices supplemented these data where economic values were unavailable. In the case of the grazing property in the WA supply chain, which produces a large mass of grain for sale as well as livestock, 100% of the on-farm GHG emissions were allocated to the livestock production system (and then further allocated to HSCW). In the feedlot, the cattle produced for the premium export market exhibited 73% of the total

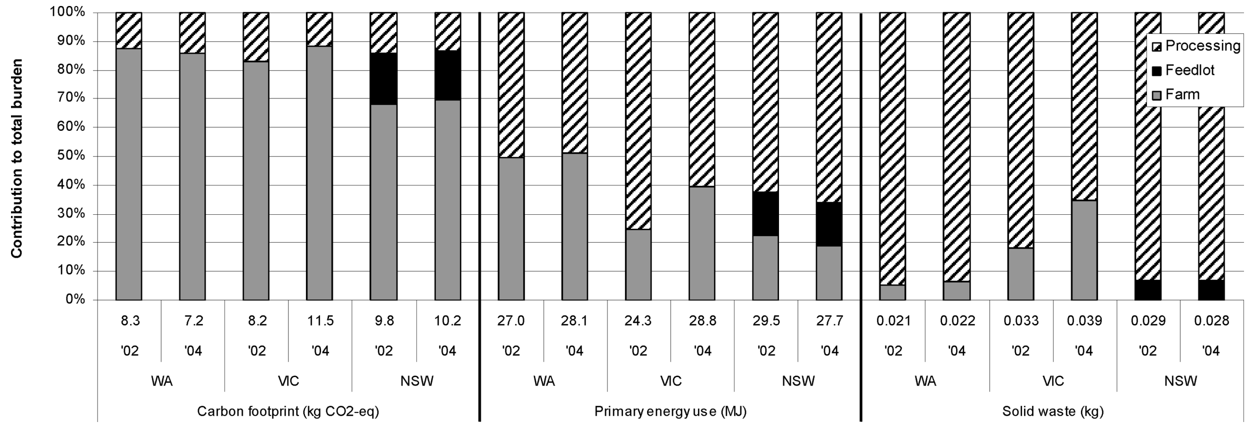

FIGURE 1. Carbon footprint (CF), primary energy use (PE), and solid waste (SW) per kilogram of HSCW by supply chain and year.

TABLE 1. Comparison of Grain- and Grass-Finished Beef

calculation for NSW supply chain, 2004	unit	grain-finished	grass-finished
HSCW gain per head at the grazing property	kg of HSCW/head	217	280
CF at the grazing property	kg of CO_2 equiv/kg of HSCW	10.6	10.6
HSCW gain per head at the feedlot	kg of HSCW/head	146	0
CF at the feedlot	kg of CO_2 equiv/kg of HSCW	5.5	0
total HSCW per head at the processor gate	kg of HSCW/head	363	280
CF at the grazing property per kilogram of HSCW at the processor	kg of CO_2 equiv/kg of HSCW	6.4	10.6
CF at the feedlot per kilogram of HSCW at the processor	kg of CO_2 equiv/kg of HSCW	2.2	
CF per kilogram of HSCW at the processor	kg of CO_2 equiv/kg of HSCW	1.4	1.4
total CF per head	kg of CO_2 equiv/head	3602	3365
total CF per kilogram of HSCW	kg of CO_2 equiv/kg of HSCW	9.9	12.0

HSCW gain on the property but consumed 76% of the feed. We assumed that the LCI would be proportional to feed consumption and allocated accordingly and then further allocated to HSCW production. Typically HSCW accounts for 55−60% of the liveweight by mass and 86−89% of the economic value. In this paper, all results are shown on a mass-allocated basis unless otherwise stated.

Results

The results for three conventional LCIA categories are presented in Figure 1. Besides the absolute result for each supply chain, year, and indicator, the graphs show the relative contribution of each meat production stage to the total impact for the respective impact category.

Carbon Footprint (CF). As expected, and in line with all other LCA work in this area, the main source of GHG emissions in the supply chains is enteric methanogenesis from the animals themselves. The data in Figure 1 show that the sheepmeat supply chain in WA performs a little better on average than the beef supply chains, although the enteric methane data (the principal GHG emission) are similar for beef and sheep when compared on the basis of HSCW gain. For example, for 2002, the average daily emission rate in the NSW supply chain was equivalent to 0.18 (kg of CH_4/head)/day for cattle that typically produce between 200 and 250 kg of HSCW on slaughter. The corresponding figures for sheep in the WA system that year were 0.02 (kg of CH_4/head)/day for animals yielding 18−21 kg of HSCW. Cattle generally stay on the grazing property longer (up to 3 years) than sheep (up to 1.5 years) before slaughter, which may partly account for the superior performance of the WA supply chain.

Nitrous oxide emissions from leguminous pastures were excluded from these calculations due to data uncertainty. For example, the NGGIC method requires three year average pasture production data which we did not have. The alternative of using a broad average from the literature (e.g., ref *45*) introduces a relatively uncertain datum: the mean value calculated on the basis of the publicly available data related to that reference is equal to its standard deviation. Nevertheless, if 35 kg of N was fixed per hectare and year (the median of those data), this suggests that the CF results for the NSW and Victorian grazing properties underestimate the total CF value by less than 10%.

In Table 1, we identify the CF from different stages of the NSW supply chain when the grass-finished and grain-finished components of that supply chain are considered separately. The comparison reveals that although the total CF per head of cattle is higher for the grain-finished beef, this is completely offset by the more efficient weight gain of the grain-finished cattle. The other main reason for the better performance of grain-fed beef with regard to GHG emissions on a per kilogram of HSCW basis is the superior digestibility of the feed and the associated reduction in methane emissions. While our results show a 38% reduction in enteric methane emissions for grain-finished beef, other studies claim reductions as great as 70% (*46*). Many potential strategies have been identified for methane abatement, and most of these options are being investigated by researchers around the world (*47*). They range from dietary manipulation (e.g., pasture and grazing management, breeding and genomics, feed additives, etc.) and animal management (e.g., earlier finishing through improved feeding or reducing unproductive animal numbers through extended lactation) to rumen manipulation (e.g., biological control and antibiotics).

It is interesting to note the interannual fluctuations in the kind of agricultural business investigated in this study. The Victorian supply chain operated as a finishing enterprise for traded cattle purchased as weaners in 2002 but included breeding activities in 2004. Consequently, the enteric methane emissions are 34% lower in 2004, but HSCW gain is 44% lower, overall contributing to a 40% higher CF per kilogram of HSCW. Similar effects are responsible for the better performance of the WA supply chain in 2004. The results also demonstrate the significance of a feedlot component in

TABLE 2. Dressing Percentages and Saleable Meat Percentages from the Literature

param	estimate	min	max	ref (estimate)	ref (min, max)
dressing percentage (beef)	53	50	62	*49*	*50*
saleable meat percentage (beef)	70	65	75	assumed from min and max	*50*
dressing percentage (sheepmeat)	47	44	50	*51*	*52, 49*
saleable meat percentage (sheepmeat)	70			*53*	

a supply chain. For example, in the NSW supply chain, where some cattle are finished on pasture and others go to a feedlot, the feedlot can be responsible for about 22% of the total CF of the whole supply chain despite the relatively short time period in which the cattle are at the feedlot compared with the grazing property.

Primary Energy Use (PE). In contrast to the GHG results, the processing plants contribute most to the total PE, mainly because of the large refrigeration equipment used. We found that larger grazing properties used more on-site energy per kilogram of HSCW produced. The other factor varying considerably is the energy demand associated with the extended supply chain. In NSW, significant expenditure on services such as cattle selling and veterinary support and materials such as fodder, medicines, and pesticides contributed to energy consumption in the extended supply chain, which represents 13−14% of the total figure. This contribution demonstrates the value of hybrid LCA compared to process analysis, although the avoided truncation error is not as high as suggested by some authors (e.g., ref *48*). All three grazing properties made significant expenditure on agricultural services and on freight, and the Victorian grazing property had significant agistment costs. In comparison to the GHG results, feed processing represented a significant contribution to the feedlot energy demand.

Solid Waste (SW). The overall SW production was dominated by the meat processing plant. Systems for collecting data on waste production at the three grazing properties were very different and rudimentary. Therefore, while the principal SW-generating activity appears to vary from property to property, we are not certain whether this reflects the underlying operations or data management in relation to waste. In the case of the WA supply chain, the principal waste-generating activity is a consequence of feed processing; in the Victorian case, it is on-site waste generation; and for the NSW supply chain, it is chemical production. These data are not accurate enough to inform policy initiatives in the area of waste minimization, but they indicate potential for improved (continual) data collection.

Soil Erosion. The soil erosion indicator approach we present is intended to enumerate environmental issues related to grazing property soils, so it refers only to the grazing property component of the red meat supply chains. The results, while built on dialogue regarding farm management practices with farmers, are enumerated using secondary (literature) data and are, therefore, less certain than the conventional LCA indicators.

The NSW property has much higher soil erosion potential, i.e., 887 (kg/ha)/yr in 2002 and 923 (kg/ha)/yr in 2004 than the WA property (13 (kg/ha)/yr in 2002; 23 (kg/ha)/yr in 2004), with the estimate for the Victorian property equal to zero. These estimates are based on the National Land and Water Resources Audit (*44*) and consider the erosion gully density and the estimated annual hillslope erosion rate. Whereas the erosion gully density is rated "low" or "very low" for all three supply chains, the hillslope erosion rate for the NSW properties is rated "low" to "very high". This reflects the soil types and the topography of the area.

Comparison of the Results. The results presented above were compared with red meat LCA results from 11 studies published between 1999 and 2008 (*4, 5, 7*−*15*). These studies were conducted in a range of different countries or regions and covered different production systems for sheepmeat and beef including conventional and organic farming principles, feedlot and pastoral production, and intensive and extensive livestock farming. Most studies' system boundaries included all farming processes, i.e., breeding, feed production and related fertilizer production and transport, farm electricity use, heating, farm field work, and waste management ("cradle-to-farm gate"). Two studies also included the environmental burdens associated with the processing and distribution stages of the meat life cycle (*4, 14*). All studies except for that of Verge et al. (*10*) exclude the environmental impacts associated with capital goods.

CF is the most commonly evaluated impact category: 10 of the 11 studies provide this indicator. While PE is assessed in 7 of the 11 studies, results for other impact categories such as eutrophication or acidification potential are rarely presented. Only one study considers different toxicity potentials (*7*); another study relates to pesticide use and abiotic resource consumption (*5*). In our comparisons we focus on CF and PE since the characterization factors for the other models represent an unknown variable.

Putting the results of different studies side-by-side, we make comparisons on the basis of unallocated burdens, except where allocation to grain and wool products occurs. The sheep farms in the literature use economic allocation to consider wool byproducts, so we present our results on the same basis for comparison.

In addition, since various FUs are used in the different studies, all had to be converted to the common basis of HSCW. This is a critical step, and sensitivity analysis shows that factors such as the dressing percentage are highly influential on the final result. The liveweight of an animal is the body mass of the animal immediately before slaughter. The carcass weight, also referred to as HSCW or dressed weight, is the liveweight multiplied by the dressing percentage, which takes into account the body parts of an animal that does not become saleable meat, i.e., the hide or fleece and the contents of the gastrointestinal tract (*49*). Boneless, retail, or saleable meat is the premium meat that is sold at the retail outlets after other parts of the animal are removed, such as bones and fat tissue. Dressing percentages and saleable meat percentages vary between countries, ages, conditions, and breeds of animals, etc. Table 2 shows the estimates we used, as well as minimum and maximum values obtained from the literature.

Table 3 illustrates the great variation in the results for both CF and PE. It should be noted that the name of the nation is used as an identifier for a particular study and does not imply representativeness of red meat production for the respective country. According to the data evaluated, beef produced in Africa in a Sahelian pastoral system has the lowest CF with 5.9 or 8.4 kg of CO_2 equiv/kg of HSCW; it is unclear whether the published result is retail or HSCW beef, so the lower value optimistically assumes the former while the higher value assumes the latter. Beef produced in Japan is 4 times more GHG-intense.

One study reports that high PE demands arise from the production of feed (grass or concentrate feeds) (*5*). This observation is consistent with the relatively high energy consumption of the WA farm examined in this study and its

TABLE 3. Comparison of CF and PE for Beef and Sheepmeat Production

area (production type)	carbon footprint (kg of CO_2 equiv/kg of HSCW) beef	carbon footprint (kg of CO_2 equiv/kg of HSCW) sheepmeat	primary energy (MJ/kg of HSCW) beef	primary energy (MJ/kg of HSCW) sheepmeat	ref
WA, 2002		10.8		47.9	
WA, 2004		10.2		42.2	
Africa (pasture, assumed boneless)	5.9				*10*
Africa (pasture, assumed HSCW)	8.4				*10*
Belgium	10.1	13.0			*13*
U.S. (feedlot)	10.4				*10*
VIC, 2002 (organic)	11.6		10.1		
NSW, 2002 (farm and feedlot)	15.0		24.4		
U.K./Wales (conventional)	15.2	17.5	26.8	23.1	*4*
NSW, 2004 (farm and feedlot)	15.4		20.0		
Sweden	15.6		18.1		*12*
U.K./Wales (organic)	17.5	10.1	17.4	18.4	*4*
VIC, 2004 (organic)	18.1		20.2		
Canada	19.6				*9*
Ireland (organic)	20.9				*11*
Ireland (agrienvironmental)	23.0				*11*
Ireland (conventional)	24.5				*11*
Japan	25.5		118.3		*8, 14*

on-farm processing of feeds for final fattening of sheep and lambs. The WA farm is, nevertheless, at the same order of magnitude as the other sheepmeat farms and an order of magnitude more efficient than the Japanese beef farm. Another study also refers to the high fossil fuel energy demand associated with ammonia production, which is the basis for N-fertilizer (*10*).

Although converting the results to a common FU and taking allocation issues into account should provide a robust basis for comparison between beef and sheepmeat production in different countries, these results must be considered with great caution since many other variables can play a major role. Our study, as well as the comparison with literature, underlines the fact that varying farm operations can significantly influence the environmental performance of red meat production. For example, our data suggest that organic production may use less energy than conventional farming practices, but may result in a higher CF. The type of animal feed can also influence the results significantly. In general terms, because beef and sheep can digest grain more easily than forage, feedlot animals tend to emit less methane than animals of the same weight on pasture (*50*). Whether the reduced enteric methane emissions from a grain diet can provide greater GHG savings than those caused by the production of the grain feed, and whether other environmental impacts from grain production such as ecotoxicity potentials will become a concern, will depend on the production system. In relation to that, the methods applied to determine enteric methane emissions vary significantly in the studies under investigation. Most studies either use IPCC standard values or apply the corresponding methodology (*36, 51*). Another study (*9*) follows a quadratic regression equation based on the dry matter intake (*52*), whereas Casey and Holden (*12*) used a nutrition software package, called RUMNUT, that is based on protein systems (*53*). Other identified variables that can affect the environmental performance associated with red meat production are (1) the lifetime of the animals (13−33 months), (2) the regime of manure management, (3) varying IPCC conversion factors for GHGs, (4) effects from the inclusion of land use change (*54*), (5) country-specific impacts from energy production, and (6) animal breed.

Considering this analysis of international studies and the fact that our work is based on three farms and two years of data, estimates of environmental consequences of red meat production should not be reported as single figures. Instead, we would advise media organizations to discuss data ranges, which are more representative of available information. The data collected in this study represent three farms in three different locations in Australia producing beef and sheepmeat over two years and can be considered a good starting point for evaluating the environmental performance of red meat production. However, further research is required to validate this information and to enable further statements about the significance and representativeness of the results of this study to be made.

Acknowledgments

This research was funded by Meat and Livestock Australia.

Note Added after ASAP Publication

There were incorrect values in Table 3 in the version of this paper published ASAP January 12, 2010; the corrected version published ASAP January 22, 2010.

Supporting Information Available

Additional basic data and assumptions. This information is available free of charge via the Internet at http://pubs.acs.org.

Literature Cited

(1) NGGIC. *National Greenhouse Gas Inventory*; Department of Climate Change, Australian Government: Canberra, Australia, 2006.

(2) Basset-Mens, C.; van der Werf, H. M. G. Scenario-based environmental assessment of farming systems: The case of pig production in France. *Agric., Ecosyst. Environ.* **2005**, *105* (1−2), 127–144.

(3) Eriksson, I. S. Environmental systems analysis of pig production—The impact of feed choice. *Int. J. Life Cycle Assess.* **2005**, *10* (2), 143–154.

(4) Schlich, E. H.; Fleissner, U. The ecology of scale: Assessment of regional energy turnover and comparison with global food. *Int. J. Life Cycle Assess.* **2005**, *10*, 219–223.

(5) Williams, A. G. E., Audsley, E. Sandars D. L. *Determining the Environmental Burden and Resource Use in the Production of Agricultural and Horticultural Commodities. Main Report*; Cranfield University and Defra: Bedford, U.K., 2006.

(6) Cederberg, C.; Mattsson, B. Life cycle assessment of milk production—A comparison of conventional and organic farming. *J. Clean. Prod.* **2000**, *8*, 49–60.

(7) Chassot, A.; Philipp, A.; Gaillard, G. Ökobilanz-vergleich von intensivem und extensivem rindfleischproduktionsverfahren: Fallstudie anhand zweier fallbeispiele (life cycle assessment

(LCA) of two beef production systems). Presented at the 8th Wissenschaftstagung Ökologischer Landbau, Ende der Nische, Kassel, Germany, March 1−4, 2005.
(8) Haas, G.; Wetterich, F.; Köpke, U. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agric., Ecosyst. Environ.* **2001**, *83*, 43–53.
(9) Ogino, A.; Kaku, K.; Osada, T.; Shimada, K. Environmental impacts of the Japanese beef-fattening system with different feeding lengths as evaluated by a life-cycle assessment method. *J. Anim. Sci.* **2004**, *72* (7), 2115–2122.
(10) Verge, X. P. C.; Dyer, J. A.; Desjardins, R. L.; Worth, D. Greenhouse gas emissions from the Canadian beef industry. *Agric. Syst.* **2008**, *98*, 126–134.
(11) Subak, S. Global environmental costs of beef production. *Environ. Econ.* **1999**, *30*, 79–91.
(12) Casey, J. W.; Holden, N. M. Greenhouse gas emissions from conventional, agri-environmental scheme, and organic Irish suckler-beef units. *J. Environ. Qual.* **2006**, *25*, 231–239.
(13) Cederberg, C.; Stadig, M. System expansion and allocation in life cycle assessment of milk and beef production. *Int. J. Life Cycle Assess.* **2003**, *8*, 350–356.
(14) Nemry, F., Theunis, J. Brechet, T. Lopez, P. *Greenhouse Gas Emissions Reduction and Material Flows*; Federal Office for Scientific, Technical and Cultural Affairs, Prime Minister's Office: Brussels, 2001.
(15) Ogino, A.; Orito, H.; Shimada, K.; Hirooka, H. Evaluating environmental impacts of the Japanese beef cow-calf system by the life cycle assessment method. *Anim. Sci. J.* **2007**, *78*, 424–432.
(16) Canals, L. M.; Domenech, X.; Rieradevall, J. Use of life cycle assessment in the procedure for the establishment of environmental criteria in the catalan eco-label of leather. *Int. J. Life Cycle Assess.* **2002**, *7* (1), 39–49.
(17) Berlin, J. Environmental life cycle assessment (LCA) of Swedish semi-hard cheese. *Int. Dairy J.* **2002**, *12*, 939–953.
(18) Blonk, H.; van Zeijts, H. Screening LCA on milkpowder. *Towards an Environmental Information Infrastructure for the Dutch Food Industry*; IVAM Environmental Research: Amsterdam, 1997.
(19) Casey, J. W.; Holden, N. M. The relationship between greenhouse gas emissions and the intensity of milk production in Ireland. *J. Environ. Qual.* **2005**, *24*, 429–436.
(20) de Boer, I. J. M. Environmental impact assessment of conventional and organic milk production. *Livest. Prod. Sci.* **2003**, *80*, 69–77.
(21) Eide, M. H. Life cycle assessment (LCA) of industrial milk production. *Int. J. Life Cycle Assess.* **2002**, *7* (2), 115–126.
(22) Hospido, A.; Moreira, T.; Feijoo, G. Simplified life cycle assessment of Galacian milk production. *Int. Dairy J.* **2003**, *13*, 783–796.
(23) Lundie, S.; Feitz, A. J.; Changsirivathanathamrong, A.; Jones, M.; Dennien, G.; Morian, M. *Evaluation of the Environmental Performance of the Australian Dairy Processing Industry Using Life Cycle Assessment*; Centre for Water and Waste Technology, The University of New South Wales: Sydney, January 2003.
(24) Narayanaswamy, V.; Altham, W.; van Berkel, R.; McGregor, M. Application of life cycle assessment to enhance eco-efficiency of grains supply chains. *4th Australian Life Cycle Assessment Conference—Sustainability Measures for Decision Support*, Sydney, Feb 23−25, 2005; Australian Life Cycle Assessment Society: Melbourne, 2005.
(25) Narayanaswamy, V.; Altham, J.; Van Berkel, R.; McGregor, M. *Environmental Life Cycle Assessment (LCA) Case Studies for Western Australian Grain Products*; Curtin University of Technology: Perth, Australia, May 2004; p 133.
(26) Wood, R.; Lenzen, M.; Dey, C.; Lundie, S. A comparative study of some environmental impacts of conventional and organic farming in Australia. *Agric. Syst.* **2007**, *89*, 324–348.
(27) Parker, D. B.; Perino, L. J.; Auvermann, B. W.; Sweeten, J. M. Water use and conservation at Texas High Plains beef cattle feedyards. *Appl. Eng. Agric.* **2000**, *16* (1), 77–82.
(28) Winchester, C. F.; Morris, M. J. Water intake rates of cattle. *J. Anim. Sci.* **1956**, *15*, 722–740.
(29) Hicks, R. B.; Owens, F. N.; Gill, D. R.; Martin, J. J.; Strasia, C. A. Water intake by feedlot steers. *Animal Science Research Report*; Agricultural Experiment Station, Oklahoma State University: Stillwater, OK, 1988; pp 208−212.
(30) Sanders, P. R.; Lott, S. C.; Watts, P. J.; Tucker, R. W.; Casey, K. D.; Lyndon, K. *Lot Fed Cattle Water Consumption*; Queensland Department of Primary Industries: Toowoomba, Queensland, Australia, July 1994.
(31) Hegarty, R. S. Strategies for mitigating methane emissions from livestock—Australian options and opportunities. *1st International Conference on Greenhouse Gases and Animal Agriculture*, Obihiro, Japan, Nov 7−11, 2001; Elsevier Science: Amsterdam, The Netherlands; pp 61−65.
(32) NGGIC. *Australian Methodology for the Estimation of GHG Emissions and Sinks 2004—Agriculture*; National Greenhouse Gas Inventory Committee, Australian Greenhouse Office: Canberra, Australia, 2004.
(33) McGrabb, G. J. Managing methane emissions from enteric fermentation. *Greenhouse Best Practice in Agriculture Workbook*; Australian Greenhouse Office: Canberra, Australia, 2005.
(34) AGO. *AGO Factors and Workbook Method*; Australian Greenhouse Office: Canberra, Australia, August 2004.
(35) Hao, X.; Chang, C.; Larney, F. J.; Travis, G. R. Greenhouse gas emissions during cattle feedlot manure composting. *J. Environ. Qual.* **2001**, *30*, 376–380.
(36) ISO. *ISO 14040: 2006 Environmental Management—Life Cycle Assessment—Principles and Framework*; International Organization for Standardization: Geneva, Switzerland, 2006.
(37) MLA. *Eco-Efficiency Manual for Meat Processing*; Meat and Livestock Australia Ltd.: Sydney, 2002; p 138.
(38) Cederberg, C. *Life Cycle Assessment of Milk Production: A Comparison of Conventional and Organic Farming*; SIK-Rapport 1998 Nr 643; The Swedish Institute for Food and Biotechnology (SIK): Gothenburg, Sweden, 1998.
(39) PE International. *GaBi Software*, version 4.2; PE International: Leinfelden-Echterdingen, Germany, 2005.
(40) Rowley, H. V.; Lundie, S.; Peters, G. M. A hybrid LCA model for comparison with conventional methodologies in Australia. *Int. J. Life Cycle Assess.* **2009**, *14* (6), 508–516.
(41) ABS. 5209.0.55.001: Australian National Accounts: Input-Output Tables−1998−99. Australian Bureau of Statistics Web site. www.abs.gov.au/AUSSTATS/abs@.nsf/DetailsPage/5209.0.55.0011998-99?OpenDocument (accessed January 2007).
(42) NGGIC. *Australian Methodology for the Estimation of GHG Emissions and Sinks 2004—Agriculture*; National Greenhouse Gas Inventory Committee, Australian Greenhouse Office: Canberra, Australia, May 2006.
(43) DCC. *National Greenhouse Accounts (NGA) Factors*; Department of Climate Change: Canberra, Australia, 2008; p 67.
(44) NLWRA. *Australian Agriculture Assessment 2001, Volume 1, Theme Report for the National Land and Water Resources Audit*; National Land and Water Resources Audit: Canberra, Australia, 2001.
(45) Unkovich, M.; Pate, J S.; Sanford, P. Nitrogen fixation by annual legumes in Australian Mediterranean agriculture. *Aust. J. Agric. Res.* **1997**, *48*, 267–293.
(46) Harper, L. A.; Denmead, O. T.; Freney, J. R.; Byers, F. M. Direct measurements of methane emissions from grazing and feedlot cattle. *J. Anim. Sci.* **1999**, *77*, 1392–1401.
(47) Eckard, R. J., Grainger, C. de Klein C. A. M. Options for the abatement of methane and nitrous oxide from ruminant production—a review. Presented at the World Animal Production Conference; Cape Town, South Africa, 2008.
(48) Lenzen, M. Errors in conventional and input-output-based life-cycle inventories. *J. Ind. Ecol.* **2000**, *4* (4), 127–148.
(49) Warriss, P. D. *Meat Science: An Introductory Text*; CABI: Oxford, U.K., 2000.
(50) IPCC. *Guidelines for National Greenhouse Gas Inventories. Agriculture, Forestry and Other Land Use*; Intergovernmental Panel on Climate Change: Cambridge, U.K., 2006.
(51) IPCC. *Climate Change 1995: The Science of Climate Change. Contribution of Working Group I to the Second Assessment Report of the IPCC*; Intergovernmental Panel on Climate Change: Cambridge, U.K., 1996.
(52) Shibata, M.; Terada, F.; Kurihara, M.; Nishida, T.; Iwasaki, K. Estimation of methane production in ruminants. *Anim. Sci. Technol.* **1993**, *64*, 790–796.
(53) Chamberlain, A. T. RUMNUT. The Ruminant Nutritional Program. Hampshire, U.K., 2004.
(54) *PAS 2050: Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services*; British Standards Institution: London, October 2008 (PAS = Publicly Available Specification).

ES901131E