

Current Approaches of Beef Cattle Systems Life Cycle Assessment: A Review

Emily M. Andreini and Sara E. Place

Department of Animal Science
Oklahoma State University

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Introduction

The U.S. beef industry has important impacts on the economy and ecosystem services, and contributes to essential societal needs for food and fiber. According to the USDA Economic Research Service (ERS) (2013), the total U.S. beef consumption in 2012 was 25.8 billion pounds, and the retail equivalent value of the U.S. beef industry was \$85 billion. As public concern about climate change and environmental sustainability continues to grow, societal pressures on the beef industry to reduce its environmental impact are rising.

Life cycle assessments (LCA) provide documentation to better understand if consumer concerns are justified by quantifying the environmental impact of beef systems and furnishing a method of identifying mitigation options suited to reduce those environmental burdens (Beauchemin and McGeough, 2013). The process of LCA is generally composed of four components: 1. Goal and scope, 2. Life cycle inventory analysis (LCI), 3. Life cycle impact assessment (LCIA), and 4. Interpretation. Creating an LCI of emissions to the environment (e.g. GHG emissions) for an animal agricultural system, such as beef cattle systems, is challenging. The diversity of management systems across operations and segments of the industry, as well as the biogeochemical processes involved, contribute to the challenges of creating LCI analyses.

The purpose of this literature review is to assess the current LCA methodologies being used in analyses of the beef industry and to compare the complexity and variation between LCA studies. This review will address areas of current LCA methodologies that are lacking in information to account for the variety of factors that affect beef production system inputs and outputs.

LCA Methodology

Life cycle assessment is essentially an environmental accounting system that sums emissions to the environment (e.g. GHG emissions) across an entire production chain of a product of interest (e.g. beef), including both direct (e.g. enteric methane emissions from cattle) and indirect (e.g. GHG emissions associated with fertilizer production) emissions sources (Rotz and Veith, 2013). The International

Organization for Standardization (ISO) has developed international standards (i.e. ISO 14040 series) for conducting LCA based on the consensus of stakeholders around the world (Finkbeiner et al., 2006). The method has been adapted from its original use in industrial operations for a wider range of applications including agriculture (Caffrey and Veal, 2013). Beauchemin and McGeough (2013) outline the four phases of a LCA as the following:

1. Goal and scope – Identifying the goal and intended use of the LCA. These considerations set the system boundary of the LCA and determine what will be included and the level of detail needed.
2. Life cycle inventory analysis (LCI) – The inventory includes the collection of input and output data necessary for the LCA.
3. Life cycle impact assessment (LCIA) – The impact assessment explains the environmental implications of the results to the environmental issue(s) of interest.
4. Interpretation – Based on the original goal, the interpretation provides conclusions and recommendations discovered in the LCA.

The four phases outlined create a basic methodology for all LCA. However, variation regarding the goal, scope, system boundaries and functional unit considered within each LCA still exist (Beauchemin and McGeough, 2013). Differences in LCA approaches generate uncertainty during the comparison of results across LCAs. In an attempt to harmonize LCA methodology in the livestock sector, the Livestock Environmental Assessment and Performance Partnership (LEAP), a collaborative effort of stakeholders from governments, non-governmental organizations, and the private sector, released draft guidelines regarding LCAs for feed supply chains, small ruminants, and poultry production in early 2014 with plans to release guidelines for large ruminants in the near future. The sector-specific methodology provides five principles that guide users through assessments of GHG emissions and other environmental impacts (LEAP, 2014). The guiding principles include relevance, completeness, consistency, accuracy and transparency (LEAP, 2014).

However, the LEAP LCA guidelines have yet to be finalized and widely adopted. The following discussion will characterize the ambiguities of LCA methods and results in the published literature for beef cattle production systems.

Considerations for LCAs of Beef Production

The wide range of activities associated with the production of agricultural products creates distinct challenges when accounting for environmental implications of each system (Caffrey and Veal, 2013). In the following sections, an overview of the major methodological, scale, and scope challenges that arise when conducting LCAs for animal agricultural products is provided.

Differences in functional units

The functional unit expresses the measure of output from the system and offers a reference point for expressing environmental impacts (Beauchemin and McGeough, 2013). The functional unit is directly related to the LCA goals and the audience the assessment will address (Caffrey and Veal, 2013). As shown in

Table 1, the functional unit used varies across LCAs of beef production systems. In order to compare differences across LCAs, the functional units used in each LCA must match to avoid inappropriate comparisons (e.g. impacts per kg of boneless

Table 1. Example of beef Life Cycle Assessments that have impacts of interest beyond greenhouse gas emissions

Reference	Location	Functional unit	System boundaries	Impacts of interest
Pelletier et al. (2010)	Upper Midwestern United States	Per kg live weight	Cradle-to-farm gate	Energy use (MJ), Ecological footprint (m ²), Greenhouse gas (GHG) emissions (CO ₂ -eq), Eutrophication (PO ₄ -eq)
Cederberg Stadig (2003)	Sweden	Per kg bone free meat	Cradle-to-farm gate	Energy use (MJ), Land use (m ²), Toxicity or pesticide use (mg active substance), Climate change (kg CO ₂ -eq), Acidification (mol H ⁺), Eutrophication (kg O ₂ -eq)
Peters et al. (2010)	Australia weight (HCW)	Per kg hot carcass	All on-site and upstream processes at the farm, feedlot and whole, processing plant, incl. transport between sites	Carbon foot (CO ₂ -eq), Primary energy use (MJ)
Nguyen et al. (2010)	European Union	Per kg meat slaughter weight delivered from farms	Cradle-to-farm gate	Global warming potential (CO ₂ -eq), Acidification (g SO ₂ -eq), Eutrophication (g NO ₃ -eq), Land occupation (m ²), Non-renewable energy use (MJ)
Casey and Holden (2006)	Ireland	Per kg live weight	Cradle-to-farm gate	Global warming potential (CO ₂ -eq)
Beauchemin et al. (2010)	Western Canada	Per kg beef	Beef production cycle: breeding stock cycle and meat production cycle	GHG emissions (CO ₂ -eq)
Ogino et al. (2004)	Japan	Per one beef animal	Beef-fattening system (8 mo of age to 26-28 months of age)	Global warming potential (CO ₂ -eq), Acidification (SO ₂ -eq), Eutrophication (PO ₄ -eq), Energy consumption (GJ)
Capper (2011)	United States	Per 1 billion kg of beef	Cradle-to-farm gate	Energy requirement (MJ), Feedstuff use (kg), Land use (ha), Water use (L), Fossil fuel energy (BTU), Manure (kg), N excretion (kg), P excretion (kg), Methane (CH ₄) (kg), Nitrous oxide (N ₂ O) (kg), Carbon footprint (kg of CO ₂ -eq)
Stackhouse-Lawson et al. (2012)	United States	Per kg of HCW	Cradle-to-farm gate	Primary GHG emissions: CO ₂ (kg), CH ₄ (kg), N ₂ O (kg), Secondary GHG emissions (CO ₂ -eq)

beef compared to impacts per kg of live-animal weight; de Vries and de Boer, 2010). The purpose of the functional unit is to define the function of the product, and as livestock serve to produce a nutritional protein source for human consumption

(de Vries and de Boer, 2010), per unit of beef (e.g. kg of beef) is preferred to per head or per hectare acre of land.

For LCAs that examine GHG emissions, further consideration must be given to the variation of different GHG's abilities to contribute to climate change. The major GHGs considered in beef production LCAs are carbon dioxide, methane, and nitrous oxide. Each of these gases has a different radiant forcing, or ability to trap heat in the Earth's atmosphere, and different atmospheric lifetime (Seinfeld and Pandis, 2006). To account for these differences, the Intergovernmental Panel on Climate Change (IPCC) has established global warming potentials (GWP) for each GHG on a carbon dioxide basis for a 100-year time horizon, often referred to as carbon dioxide equivalents (CO₂-eq). The 100-year GWP from the 2007 IPCC report are 1 for carbon dioxide, 25 for methane, and 298 for nitrous oxide, meaning methane and nitrous oxide have a 25 and 298 times greater potential, respectively, to trap heat on a 100-year time horizon when compared to carbon dioxide (Solomon et al., 2007). The 2013 IPCC report revised 100-year GWPs for methane and nitrous oxide to 28 and 265, respectively, without the inclusion of climate-carbon feedbacks and 34 and 298, respectively, with the inclusion of climate-carbon feedbacks (IPCC, 2013).

While the standardization of GWP exists, consideration must be given to the GWPs used by each LCA, as different LCAs in the literature use different GWPs, which can influence the CO₂-eq per unit of beef reported. For example, Casey and Holden (2006) report CO₂-eq emissions per kg of live weight using GWP of 21 for methane and 310 for nitrous oxide, while Foley et al. (2011) reported CO₂-eq emissions per kg of carcass weight using a GWP of 25 and 298 for methane and nitrous oxide, respectively. Thus, conversions of the functional units and CO₂-eq to a common scale would be required before comparing the results between these two partial LCAs of Irish beef production systems.

Differences in geographic locations

It is difficult to define an average system as the management of livestock fluctuates across farms, and there is a wide range of production systems

according to topography, calving season, and type of diet (Wilkinson, 2011). Direct comparison between regional and national LCA results may not be appropriate due to differences in cultivation practices and climates (Caffrey and Veal, 2013).

In a report by the Food and Agriculture Organization (FAO) of the United Nations, titled *Livestock's Long Shadow* (LLS), global numbers were inappropriately extrapolated by some in the media to regional levels and applied out of context (Pitesky et al., 2009). LLS attributes 18% of the global anthropogenic GHGs to livestock production and concluded that the livestock sector produces a higher share of anthropogenic GHG emissions than the transportation sector (Steinfeld et al., 2006). The comparison of the livestock and transportation sectors was questioned by some due to the differences in methodologies used to determine the contributions of the respective sectors to total anthropogenic GHG emissions (i.e. a LCA was not conducted for global transportation; Pitesky et al., 2009). Inappropriate prediction and attribution of GHG emissions to regional, national or global scales can cause confusion and an inaccurate interpretation of results; therefore, disaggregating LCA results by region or nation rather than reporting global averages, can be particularly useful.

Comparisons across production systems in different geographic locations require consistent boundaries, goals and functional units used in LCAs for each location. As a follow-up to LLS, Opio et al. (2013) conducted a global LCA of GHG emissions from ruminant production systems worldwide and disaggregated their analysis and results by geographic region. Using consistent methodology, scope, and scale, Opio et al. (2013) found GHG emissions per carcass weight for cattle ranged from 14 kg of CO₂-eq/kg of carcass weight in Eastern Europe and the Russian Federation to 76 kg of CO₂-eq/kg of carcass weight in South Asia. Differences across regions in emission intensity can be attributed to production practices and the relative contribution of dairy cattle to the beef supply (discussed further in "Allocation" section).

Differences in system boundaries

The system boundaries of the LCA outline the processes that will be assessed, and the goal and scope of the study will determine the limits of the LCA.

The system boundary defines all areas of the system that are included or excluded in the LCA (Crosson, et al., 2011). Most LCAs use a system boundary that ends at the farm gate (sometimes referred to as partial LCA), as most GHG emissions from meat and milk are produced up to the time the product leaves the farm (Beauchemin and McGeough, 2013). When the LCA boundary is beyond the farm gate, end-product processes such as food processing, transportation and post-harvest handling are also included (Caffrey and Veal, 2013). Inconsistency in the system boundary can create difficulty in comparing studies and interpreting results as demonstrated in **Table 1**, which outlines the variation of system boundaries across published LCAs of beef production systems.

Additionally, the level of analysis considered can greatly influence conclusions. Van Middelaar et al. (2013) examined the influence of the level of analysis (i.e. animal, farm, chain) on the effect of dietary manipulation on methane and other GHG emissions, and found that results are highly dependent on the level of analysis selected. At the animal level, increasing maize silage at the expense of grass and grass silage showed an immediate effect on GHG emissions and appeared to be a viable mitigation strategy (Van Middelaar et al., 2013). However, the same approach was not feasible at the farm and chain levels due to EU regulation preventing further grassland reductions, and the number of years it would take for the reduction in annual emissions to compensate for emissions from land use change, respectively (Van Middelaar et al., 2013).

Allocation: Integration with the dairy and crop industries

Dairy

Another challenging issue with beef production LCA and system boundaries concerns integration with the dairy industry. In major dairy producing states (particularly in the Western United States), male Holstein calves often enter the beef production chain by first being fed at calf ranches, and then entering beef feedlots. Additionally, culled dairy cows enter the beef supply chain once they are removed from the herd. Approximately 16.3% of cattle slaughtered in the United States originate in the dairy industry (USDA, 2013). Differences across LCA regarding how environmental impacts from the two industries are allocated further complicate comparisons. Stackhouse-Lawson et al. (2012) simulated beef

production systems with the Integrated Farm Systems Model (IFSM; USDA-ARS, 2013) to assess the carbon footprint of beef production systems in California. To integrate dairy cattle into the assessment, Stackhouse-Lawson et al. (2012) accounted for all sectors of the beef and dairy industries that contributed to meat production in the state, which included cow-calf, stocker and feedlot phase for Angus beef production and the calf ranch and feedlot phase of Holstein steer management. Production systems were defined through consultation and representative farms or ranches were simulated for each phase of the production system for beef and Holstein calves. Stackhouse-Lawson et al. (2012) reported simulated emissions from an Angus beef production system including a stocker phase as 22.6 ± 2.0 kg CO₂-eq/kg hot carcass weight (HCW), simulated emissions from an Angus beef production system without a stocker phase as 21.2 ± 2.0 kg CO₂-eq/kg HCW, and simulated emissions from a Holstein beef production system as 10.7 ± 1.4 kg CO₂-eq/kg HCW. The Holstein beef production system did not include the breeding stock, as the emissions from dairy cows would be allocated to dairy production, which resulted in the Holstein production system contributing 50 to 63% fewer CO₂-eq emissions compared to the Angus beef system (Stackhouse-Lawson et al., 2012). The results reported by Stackhouse-Lawson et al. (2012) attributed 68 to 72% of the total GHG emissions to the cow-calf phase, 14% to the stocker phase and 17 to 27% to the Angus feedlot phase of California beef production. The cow-calf phase of Angus beef production was the major contributor of GHG emissions compared to the major GHG contribution from Holstein beef production, which was the feedlot phase (Stackhouse-Lawson et al., 2012). Emissions from Holstein beef production should not be double-accounted for in dairy production, which is important to consider when conducting LCAs for coupled animal production systems (Stackhouse-Lawson et al., 2012).

Capper (2011) defined three major sections of the U.S. beef industry, similar to Stackhouse-Lawson et al. (2012), which included the cow-calf unit, the stocker/backgrounder operation, and the feedlot (including calf-fed beef and dairy animals that enter the feedlot at weaning, and yearling-fed beef animals that enter after the stocker stage). To integrate the inputs from the dairy industry into the beef industry, Capper (2011)

calculated resource inputs and waste outputs between the dairy and beef systems based upon a biological allocation method. Deterministic models of resource use and environmental impact for dairy and beef production were based upon the same nutrition and metabolism principles (Capper et al., 2009; Capper, 2011). Capper (2011) attributed the environmental cost of dairy cull cows entering beef production by using the dairy model described by Capper et al. (2009) to determine the proportion of total resource inputs and waste outputs attributable to growing Holstein heifers from birth to weight sold as beef animals. To account for the environmental cost of calf-fed dairy animals entering the feedlot phase, Capper (2011) separated the proportion of total resource inputs and waste outputs attributable to pregnancy in lactating and dry dairy cows, and adjusted the environmental cost accordingly for the number of cows required to produce the number of dairy calves in the beef system. Capper (2011) found that 4% of the total carbon footprint per unit of beef was from dairy production using the allocation methods outlined above. The percent of the environmental burden of beef production that comes from dairy production systems will vary based on allocation method (Capper, 2011) and should be considered for LCA comparisons.

Cederberg and Stadig (2003) examined LCA allocation methods of dividing the environmental burden of milk and beef production systems among products, where the core system assessed was milk production and the extended system was beef production. Four allocation methods were analyzed by Cederberg and Stadig (2003) including:

1. No allocation – all environmental burdens were attributed to milk.
2. Economic allocation – environmental burdens were distributed based on the Swedish Dairy Association's average income calculations of the products as 92% for milk, 6% for meat from cull cows, and 2% for meat from surplus calves.
3. Cause-effect physical ('biological') allocation – environmental burdens were divided based on the proportion of feed intake required to produce milk by the Swedish feeding recommendations (allocates 85% to milk and 15% to meat from cull cows and surplus calves).

4. System expansion – avoided allocation by expanding the system to consider meat from culled cows and surplus calves as products entering the beef production system, and environmental burdens from those animals were displaced to the beef system.

The 'no allocation' method exhibited the highest amount of GHG emissions attributed to milk production, and barely two-thirds of the emissions were attributed to milk production when 'system expansion' was used to avoid allocation (Cederberg and Stadig, 2003); therefore, the allocation method used affects distribution of the environmental burden and the results of the LCAs (Cederberg and Stadig, 2003). Cederberg and Stadig (2003) provide allocation methods that can be used in LCAs for co-product integration, and the variation of results further exemplifies the requirement of allocation standardization for LCA comparisons.

A recent dairy LCA by Thoma et al. (2013) used a co-product allocation method similar to the biological or cause-effect physical allocation method described in Cederberg and Stadig (2003) for milk and beef products produced by dairy production. The allocation method uses the ratio of feed energy input for meat and milk products to distribute environmental burdens. The largest input into the production of milk is the dairy feed ration consumed, but the feed net energy conversion efficiency to milk and to meat is not the same (Thoma et al., 2013). Thoma et al. (2013) allocated GHG emissions to the beef industry by allocating GHG emissions associated with the feed used for gain (e.g. enteric methane, soil nitrous oxide emissions) of animals culled for beef to the beef sector. The culled animals included calves and dairy cows; however, any impacts for additional gain once the animal left the dairy system (i.e. the GHG emissions associated with growing a Holstein steer to slaughter weight in a feedlot system) were not accounted for in the LCA for milk.

As the examples above demonstrate, there is significant variability across published full and partial LCAs regarding the allocation of environmental impacts between the beef and dairy sectors; therefore, comparisons of existing LCAs should take into account the different allocation methods.

Additionally, the variability of the year-to-year contributions of the dairy industry to beef production (driven by year-to-year changes in economic conditions for the beef and dairy industries), and uncertainties of different allocation methods should be quantified and reported.

Feed Co-products

In addition to the dairy-beef integration, another major issue of allocation of environmental impacts or burdens during LCA arises when considering co-product feeds (e.g. distiller's grains from ethanol production). Pelletier et al. (2010) conducted a LCA of three beef production systems in the Upper Midwest of the United States and allocated emissions and resource use due to crop co-products using the gross chemical energy content of co-product streams entering the beef production system. Pelletier et al. (2010) chose this allocation method based on the knowledge that caloric energy is the main driver of food production, and the chemical energy of products present in raw materials distributed between processed outputs is proportional to the system's efficiency to provide food energy. Co-products can result from the production of protein concentrate feeds used in beef production, as is the case with soybean meal and soy oil (Nguyen et al., 2010). Rather than allocating impacts between soybean meal and soy oil, Nguyen et al. (2010) utilized system expansion to estimate the environmental impacts of soybean meal due to the need for protein feed in livestock production, meaning all of the impacts due to the production of the soybeans were allocated to the soybean meal. Capper (2011) avoided accounting for feed co-products, but highlighted that these feeds (e.g. distiller's grains, beet pulp, citrus pulp, potatoes) are highly regionalized in their use and not accounting for their use could potentially overestimate land use requirements of the beef industry. Rotz et al. (2013) simulated the U.S. Meat Animal Research Center (MARC; Clay Center, NE) with the Integrated Farm System Model (IFSM) and based the energy use and CO₂-eq emissions for wet distillers grains prior to entering the beef production chain on the feed products the distillers would replace (corn and urea). While energy for transport was accounted for in their estimate, Rotz et al. (2013) did not account for the energy required for the distillation process. The current draft feed supply chain guidelines from

LEAP (2014) specify distillers grains from ethanol production as a residue, and therefore, no upstream impacts (e.g. from the production and harvest of corn) should be allocated to livestock. However, the guidelines do specify that any further processing of the residues to make them more suitable for livestock production (e.g. drying) should be allocated to livestock production (LEAP, 2014). As with dairy-beef allocation, the examples above illustrate that significant variability exists in the published literature as to how the impacts of feed co-products are allocated. Efforts such as LEAP should be useful in creating consensus regarding issues of allocation and add clarity to beef production LCA methods.

Environmental impacts beyond greenhouse gases

For livestock production systems, there are numerous environmental impacts of interest beyond simply livestock's contribution to climate change, including acidification potential, eutrophication potential, abiotic depletion, desiccation, odor, water resources and land competition (Beauchemin and McGeough, 2013). Many previous LCAs have fixated on GHG emissions, but there is an increasing interest and need to focus on multiple impact categories beyond GHG emissions. **Table 1** lists partial LCA studies that have incorporated environmental impacts and resource uses beyond GHG emissions. Due to enteric methane produced through ruminal fermentation, ruminants are a significant source of GHG emissions, but grazing ruminants also provide environmental benefits and ecosystem services (Beauchemin and McGeough, 2013). Ecosystem services may include beneficial preservation of habitats for wildlife and improved biodiversity with grazing systems (Del Prado et al., 2013). Additionally, GHG mitigation practices may have the unintended consequence of increasing the environmental impact of beef production in another environmental impact category; therefore, focusing solely on one impact (i.e. GHG emissions) may provide an incomplete environmental analysis if LCA is used in public policy making. Although the inclusion of other environmental impacts of interest complicates LCAs further, there is a need to expand beyond GHG emissions to better address sustainability (i.e. the social, economic, and environmental impacts and benefits of beef production).

Methodologies for Estimating Environmental Impacts from Beef Production Systems

The prediction of GHG emissions can vary depending on the model or prediction equation used in the LCA. Empirical, statistical methods of estimating emissions (based on emission factors) versus integrated, dynamic models that allow for interaction between emissions sources will produce differing results for assessment. A simple empirical, statistical method may multiply a number of cattle by an emission factor to estimate enteric and manure methane emissions from cattle, as is the case for IPCC (2006) Tier 1 methodology for estimating enteric methane emissions. Alternatively, other LCAs have relied upon other published estimates of beef production impacts to the farm gate. Sanders and Webber (2014) estimated the agricultural production phase (i.e. GHG emissions to the farm-gate) of a full LCA of the U.S. beef supply chain by using the weighted average CO₂-eq emissions of the whole-farm systems models reported in the review of Crosson et al. (2011), rather than attempting to build an emission inventory with emission factors.

Others have used whole-farm simulation models that are either empirical, process-based, or a mixture of empirical and process-based for partial LCAs of beef production systems. Empirical models rely on relationships or correlations derived from experimental data, while process-based models (sometimes referred to as mechanistic or biogeochemical models) attempt to model a system by dividing it into its individual components and analyzing the connections and interactions between the components (France and Kebreab, 2008). HOLOS is an example of an empirical virtual farm simulation model that relies primarily on IPCC (2006) methodologies to predict GHG emissions from soil, enteric fermentation, manure, on-farm energy use, and off-farm energy use for manufacturing fertilizers and herbicides (Beauchemin et al., 2010). While the integration of the cropping and animal production components is currently somewhat limited in HOLOS, efforts are underway to improve interactions across components and better track carbon flows (Kröbel et al., 2013).

Other process-based models, such as the Manure-DNDC (Manure Denitrification Decomposition) model, attempt to model the underlying processes and biogeochemistry that drives emissions from agricultural production (Li et al., 2012). Currently, Manure-DNDC does not have the same level of dynamic, mechanistic detail integrated into the animal production portion of the model as exists in the soil and manure storage components of the model; however, the plan is to remedy those shortcomings in further stages of development (Li et al., 2012). IFSM is an example of a model that combines empirical and process-based modeling techniques to simulate whole-farm systems, from the cropping requirements to feed cattle a given diet, to predicting enteric methane emissions based on the diet composition and intake, to simulating manure-derived methane, ammonia, and nitrous oxide emissions based on the predicted nutrient excretion and environmental conditions (USDA-ARS, 2013). By simulating the major farm components on a process level, and following nutrients and resource use through the different components of the system, IFSM allows for the tracking of carbon, phosphorus, and nitrogen flows throughout the whole farming system. Additionally, IFSM simulates production costs along with environmental impacts and resource use. Evaluation of IFSM with U.S. MARC data found little difference between simulated and actual data for the year 2011, with the difference between actual and simulated alfalfa and grass hay production, natural gas use, and fertilizer costs at 0.6%, 0.3%, and 0.5%, respectively (Rotz et al., 2013). Dynamic simulation models can be combined to produce the needed inputs and outputs required for LCAs, and may be able to overcome the issues of finding representative data for beef production LCAs (Pelletier et al., 2010). By simulating a specific production system and taking into account the variability driven by weather differences year-to-year, it may be possible to reduce the input error compared to LCAs not coupled with dynamic, process-based models (Rotz and Veith, 2013).

The following sections will provide a brief overview of the variety of methods used to estimate primary emissions from three of the largest emissions sources in the pre-farm gate beef production chain: enteric methane, manure, and soil and cropland sources.

Enteric methane

Enteric methane is the single largest GHG emissions source in beef production LCAs and most of the enteric methane emissions come from the cow-calf sector of the industry where animals are primarily grazing. For example, Beauchemin et al. (2010) attributed 63% of total CO₂-eq emissions in a cradle-to-farm gate LCA to enteric methane emissions, and 84% of the enteric methane emissions to the cow-calf system. As a consequence, the accuracy and precision of enteric methane emissions predictions can have a significant impact on beef production LCAs. **Table 2** provides examples of common prediction models for enteric methane emissions in beef production LCAs. The use of mathematical models to predict methane can help avoid extensive and costly experiments requiring cattle (Ellis et al., 2007). Mathematical models can be classified as statistical models (e.g. relate nutrient intake to methane production directly) or dynamic mechanistic models (e.g. estimate methane production using mathematical descriptions of rumen fermentation biochemistry; Kebreab et al., 2006). Numerous LCAs predict enteric methane emissions using the IPCC

(2006) Tier 2 methodology, including Nguyen et al. (2010), Pelletier et al. (2010), and Casey and Holden (2006). Ellis et al. (2010) evaluated several enteric methane emissions prediction models used in dairy whole-farm system models, including the IPCC Tier 2 methodology, with enteric methane emission measurement data from live-animal experiments, and found that overall the models had low prediction accuracy. All models evaluated had particular difficulty predicting the wide range in enteric methane emissions observed in live-animal experiments, which are affected by diet type and level of intake (Ellis et al., 2010). Moraes et al. (2014) developed new enteric methane prediction equations using a dataset from 2,574 indirect calorimetry records from both beef and dairy cattle, and compared the newly developed models to IPCC Tier 2 and FAO (2010) methodology. Across all classes of animals (heifers, lactating cows, dry cows, and steers), the authors found their new models that use gross energy intake as the only user required input consistently outperformed both the IPCC Tier 2 and FAO models (Moraes et al., 2014). As a caveat, the dataset did not include any emissions from grazing cattle nor did the diets fed to the steers

Table 2. Examples of enteric methane prediction equations commonly used in beef Life Cycle Assessments (LCA)

Model	Reference(s)	Enteric methane (CH ₄) emissions prediction equation
IPCC Tier 2	Casey and Holden (2006); Nguyen et al. (2010); Pelletier et al. (2010); Beauchemin et al. (2010);	$CH_4/\text{head}/\text{yr} (\text{kg}) = (\text{GE} \times Y_m \times 365 \text{ days}/\text{yr}) / (55.65 \text{ MJ}/\text{kg} \text{ CH}_4)$
Shibata et al. (1993)	Ogino et al. (2004)	$CH_4 (L/d) = -17.766 + 42.793 \times (\text{kg DMI}/d) - 0.849 \times (\text{kg DMI}/d)^2$
Mills et al. (2003)	Stackhouse-Lawson et al. (2012)	$CH_4/\text{head}/\text{day} (\text{kg}) = [E_{\text{max}} - E_{\text{max}}(-c^f \times M_{\text{EI}}^g)] \times F_{\text{kgCH}_4}$
Blaxter and Clapperton (1965)	Peters et al. (2010), grazing emissions	$CH_4/\text{head}/\text{day} (\text{kg}) = 1.3 + 0.112\text{DMD}_{\text{ijkl}}^i + L_{\text{ijkl}}^j (2.37 - 0.050\text{DMD}_{\text{ijkl}})$
Moe and Tyrrell (1979)	Capper (2011); Peters et al. (2010), feedlot emissions	$CH_4/\text{head}/\text{day} (\text{MJ}) = 3.406 + 0.510 \text{NFC}^k + 1.736 \text{HC}^l + 2.648 \text{C}^m$

^aGE = gross energy intake, MJ/head/day

^b Y_m = methane conversion factor, percent of gross energy in feed converted to methane

^c55.65 (MJ/kg CH₄) is the percentage of gross dietary energy lost as methane

^dDry matter intake

^e E_{max} = maximum possible emission, 45.98 MJ CH₄/head/day

^f $c = -0.0011 \cdot [\text{Starch}/\text{ADF}] + 0.0045$, Where: Starch = starch content of diet, ADF = acid detergent fiber content of diet

^gMEI = metabolizable energy intake, MJ/head/day

^h F_{kgCH_4} = conversion of MJ to kg of CH₄, 0.018 kg CH₄/MJ

ⁱ DMD_{ijkl} = digestibility of feed (expressed as a %)

^j L_{ijkl} = feed intake relative to that needed for maintenance

^kNFC = nonfiber carbohydrate (kg/d)

^lHC = hemicellulose (kg/d)

^mC = cellulose (kg/d)

contain the level of concentrates found in most feedlot cattle diets today (Moraes et al., 2014). The difficulty in collecting grazing cattle methane emissions and the consequential relative dearth of enteric methane emission data from grazing cattle in the literature is troubling for enteric methane prediction equations. Without datasets to evaluate models, little can be known about the accuracy and precision of commonly used enteric methane prediction equations for grazing cattle, which as discussed above, typically represents the largest share of CO₂-eq emissions in beef production LCA. Continued improvement of enteric methane emission equations for grazing and feedlot cattle is required for increasing the accuracy and reducing the uncertainty of LCA

GHG emissions estimates, and LCA practitioners should take care when selecting models for predicting enteric methane emissions.

Emissions from manure

Manure from cow-calf, stocker, and grass-finishing systems may be deposited directly to the pasture or rangeland where the cattle are located, while manure in feedlot systems will be more concentrated and likely managed in some way (e.g. composted); thus, emissions from these systems will vary. Furthermore, manure can be collected from confined pens and utilized as fertilizer for various crops or pastures, whereas, for cattle on pasture or rangeland, the manure remains where it was deposited (Beauchemin et al., 2010). Typically, methane emissions from manure in pastures or solid manure in feedlots are relatively minor due to the lack of an anaerobic environment, which is required for methanogenesis. In the cradle-to-farm gate LCA of a Western Canadian beef production system by Beauchemin et al. (2010), IPCC (2006) methodology was used in conjunction with NRC (2000) equations (which were used to predict N excretion in manure based on the animal dry matter intake and dietary crude protein) to estimate nitrous oxide emissions derived from manure. Considering both feedlot and cow-calf manure emissions, 23% of nitrous oxide emissions were attributed to manure with 3% of emissions from the feedlot system and 20% of emissions from the cow-calf system (Beauchemin et al., 2010). In addition to Beauchemin et al. (2010), IPCC (2006) methodology has been used to predict manure nitrous oxide emissions by Capper et al. (2011), Nguyen et al. (2010), Pelletier et al. (2010), Peters et al. (2010), and Stackhouse-Lawson et al. (2012).

The Pelletier et al. (2010) partial LCA also predicted ammonia emissions to the environment from manure using IPCC (2006) methodology, while Stackhouse et al. (2012) and Rotz et al. (2013) predicted ammonia emissions from different sources (i.e. manure in feedlots vs. manure on rangeland or pastures) with different functions including air velocity, temperature, surface pH, ammoniacal-N concentration, exposed surface area, and precipitation. For both nitrous oxide and ammonia emissions from manure, considerable temporal and spatial variability exists in emissions due to the underlying biogeochemical processes that

are responsible for emissions. Further refinement and evaluation of process-based models, such as Manure-DNDC, may improve the accuracy and precision of LCA estimates if they are integrated into LCA methodology to generate emissions inventories (Li et al., 2012).

Soil and cropping systems

Soil and cropping systems are inherently linked to the animal production component in beef production systems, due to the use of manure as fertilizer in many production systems and the interrelatedness of cropping management decisions on feed availability and quality to the animals. Cattle manure can provide nutrients, improve soil structure and increase vegetative cover to reduce potential soil erosion; however, manure applied to soils exceeding crop nutrient requirements can have negative environmental impacts (Knowlton and Ray, 2013). Nitrous oxide emissions, phosphorus runoff, nitrate leaching, and soil carbon sequestration are determined by the interactions between management, soil type and microbial populations, animal and plant genetics, and weather conditions within the production system (Del Prado et al., 2013). Consequentially, as with emissions from manure, biogeochemical models integrated with LCAs would likely improve the accuracy and precision of input and output inventories required for LCAs.

Typically, carbon dioxide sequestration by plants in the beef production system LCA is ignored due to the assumption that all the carbon sequestered by the plants will be respired by the animals eating the plants; therefore, both carbon dioxide sequestration and respiratory carbon dioxide are typically ignored (Steinfeld et al., 2006). However, the partial beef LCAs that have used IFSM (e.g. Stackhouse-Lawson et al., 2012; Rotz et al., 2013) to estimate emissions have accounted for biogenic carbon dioxide flows, including the assimilation of carbon into crops and animal growth, the respiration emissions from animals and manure, and the conversion of the carbon dioxide assimilated by crops into methane once consumed by the animal. For Stackhouse-Lawson et al. (2012) accounting for biogenic carbon reduced the simulated carbon footprint of a simulated California Angus beef production system from 22.6 to 17.7 kg of CO₂-eq/kg of HCW. Thus, while

often ignored in most beef production LCAs, biogenic carbon flows between soil and cropping systems and the animal enterprise, may warrant further consideration in future LCAs; however, integrating biogenic carbon flows into LCA methodology will require whole-farm modeling systems capable of tracking carbon.

Production Efficiency Considerations

By 2050, the world population is expected to reach 9.6 billion and meat demand is expected to grow by 73% relative to 2010 levels (Gerber et al., 2013). In order to responsibly meet the protein demands of a growing population of increasing affluence, the animal industries will need to improve productive efficiency while limiting environmental impacts. Production efficiency in the beef industry can be defined as decreasing the resource inputs and waste outputs (e.g. feed, fossil fuels, and GHG emissions) required to produce one unit of beef and can be influenced by a number of factors including genetics, reproductive efficiency, biotechnology, nutrition, animal health and welfare. Consideration of these topics in LCA methodology could improve the accuracy of LCAs to predict the environmental burdens of production systems. Additionally, due to regional, breed, and management differences across U.S. beef operations, particularly in the cow-calf sector, variation in operation-to-operation production efficiency may introduce significant uncertainty and error in LCAs of U.S. beef production. Inability to capture across operation and temporal variability from pre-farm gate impacts is an inherent limitation of LCAs that are a snapshot in time and do not attempt to account for regional differences. LCA has been adopted from industrial processes, which have an accuracy advantage when estimating inputs and outputs compared to the biogeochemical processes involved in beef production that are responsible for the bulk of the environmental emissions in the pre-farm gate portion of the supply chain. Therefore, further research building and linking dynamic, process-based models estimating pre-farm gate emissions to the LCA methodology could improve the accuracy of results and provide more useful estimates of uncertainty. To that end, the following sections provide examples of the components of the beef production system that can impact production

efficiency and should be incorporated and evaluated using LCA methodology.

Genetics

Vast genetic improvements have increased productivity in beef cattle systems and reduced environmental impacts per unit of production (Capper et al., 2011). Statistical methods for accurately calculating the estimated breeding value (EBV) of animals have shown great success, and the development of genomic selection (the use of DNA markers spread throughout the genome to track sections of DNA associated with trait variation) could provide an accurate method of predicting EBV earlier in the life of an animal, thereby increasing the rate of genetic change in the beef cattle population (Van Eenennaam, 2013). The development of genomic selection prediction equations could be beneficial in identifying traits related to the production of environmental burdens, such as enteric methane emissions and feed efficiency (Van Eenennaam, 2013). Improved feed efficiency is associated with cost reduction; furthermore, it may be associated with decreasing the environmental impacts of beef production. Ranked by residual feed intake (RFI), Nkrumah et al. (2006) examined the relationship of feed efficiency in feedlot cattle, with performance, energy partitioning, and measured oxygen consumption and methane production using an open-circuit, indirect calorimetry system. Nkrumah et al. (2006) reported 28% less methane production in low-RFI animals compared to high-RFI animals. Although the mechanisms driving the observed differences in methane emissions among animals, independent of feed intake, is uncertain, Nkrumah et al. (2006) determined high-RFI animals with increased methane production had decreased energetic efficiency and increased impacts on the environment. Selection for cattle with increased feed efficiency could impact life cycle emissions through reducing feed resources required per unit of beef and reducing enteric methane per unit of beef. Comparing beef production systems that have continuously improved genetic merit over generations to production systems that have not made genetic progress over time could provide further insights into the influence that genetic improvements have on life cycle impacts of beef

production via production efficiency. Developing methodology that includes the impacts of genetic changes over time due to selection and emphasis on production and functional traits may lead to more advanced LCAs that better capture the dynamic nature of the beef cattle production systems.

Reproductive efficiency

The improvement of reproductive efficiency has the potential to benefit the economic and environmental impacts of beef production through increasing the percent of cows that produce a calf each year. In the United States, the use of management tools to improve reproductive efficiency is highly variable, with only 11.3% of cow-calf producers with herd sizes of 1-49 cows using palpation and/or an ultrasound for pregnancy diagnosis compared to 71.7% of cow-calf producers with herd sizes of 200 or more cows (USDA, 2009). Thus, while using representative reproductive efficiency measures in beef production LCAs is typical, much of the variation that exists in cow-calf systems may not be accounted for. Increasing calving rates has the potential to increase production efficiency and reduce environmental impacts via decreasing the total maintenance costs of the beef cattle herd (e.g. decreasing the number of open cows consuming resources that are not contributing to the overall production of beef). Additionally, improved cow fertility reduces the number of replacement heifers required to maintain the same beef productivity level of the beef system, thereby decreasing environmental impacts (O'Brien et al., 2014). Because heifers are consuming feed and producing GHG emissions before they reach calving age, increasing heifer development and lowering age-at-first calving can increase production efficiency and decrease the amount of GHG emissions per unit of beef. Beauchemin et al. (2010), found the cow-calf system accounted for 80% of GHG emissions, and 24% of the cow-calf emissions were produced by breeding stock that do not immediately produce beef (e.g. growing heifers). As the work of Beauchemin et al. (2010) demonstrates, some partial beef LCAs have already considered reproductive efficiency in their methodology, and future beef production LCAs should include measures of reproductive efficiency with consideration that significant regional differences in reproductive efficiency can exist due to climate, breed, and management.

Biotechnology

Biotechnology use in the beef industry, such as growth-enhancing technologies (GET), can improve the efficiency of beef production systems. In live-animal experiments conducted by Coopridge et al. (2011) and Stackhouse-Lawson et al. (2013), the use of biotechnologies increased growth rates and decreased environmental impacts per unit of beef. Furthermore, Capper and Hayes (2012) evaluated the environmental and economic impacts of removing GET from beef production through the use of a deterministic model in a partial LCA. The GET evaluated by Capper and Hayes (2012) included steroid implants, ionophores, melengestrol acetate, and beta-adrenergic agonists. The use of GET in a production system improves growth rates and reduces the resources required to produce an equivalent amount of product compared to a system not using the technologies (Capper and Hayes, 2012). Therefore, while some LCAs have accounted for the impacts of GET on beef LCA emissions (e.g. Pelletier et al., 2010; Stackhouse-Lawson et al., 2012), future LCA practitioners should consider the impacts and the extent of GET use in the beef production systems they are evaluating.

Dynamic nature of stocker sector from year to year

The stocker industry in the United States fluctuates depending on forage availability, and the prices of cattle and other commodities (e.g. corn and soybeans). The stocker-cattle segment provides the beef industry with immunocompetent weaned feeder cattle that are accustomed to feed bunks and water sources, grouped in load lots and ready to enter a feedlot (Beck et al., 2013). Although cycling cattle through the stocker phase of beef production before entering a feedlot appears ideal, influencing factors on forage growing conditions (e.g. drought) may deem land unavailable for stocker production. Animal performance in the stocker industry is determined by grazing season and forage species, which can vary greatly from year-to-year based on climate (Beck et al., 2013). Previous LCAs have accounted for the stocker phase (e.g. Pelletier et al., 2010; Stackhouse-Lawson et al., 2012); however, including a range of stocker cattle population sizes in future LCAs linked to explanatory factors, such as forage availability and economics, could improve LCAs of beef production.

Nutrition

The nutrition of cattle has major implications for the production efficiency of beef both from the perspective of the natural resources required and the environmental emissions generated per unit of beef. As discussed previously, enteric methane emissions are greatly influenced by diet, and prediction equations for enteric methane emissions often have difficulty representing the variability observed in live-animal experiments. Numerous dietary factors influence the production of methane including the level of feed intake, type of carbohydrate, and forage processing (Johnson and Johnson, 1995), with cattle fed grain-based diets emitting fewer enteric methane emissions compared to cattle fed forage-based diets. Additionally, the nutrition of cattle greatly determines the animals' manure-nutrient excretion and productivity, both of which are important drivers of environmental and economic impacts per unit of beef. Moraes et al. (2012) evaluated the impact of reducing enteric methane emissions from dairy cow diets, and found that decreasing enteric methane emissions through dietary strategies alone increased feed costs and nitrogen and potassium excretion in the manure. Furthermore, variability in the diets fed to cattle affects crop production and the environmental impact of the cropping system. Therefore, the impacts of potential enteric methane emissions mitigation strategies should be analyzed with LCA methodology to determine if changes in cropping systems or other feed sourcing challenges (e.g. expanded use of dietary fats, like coconut oil, to reduce enteric methane) may offset benefits to reducing enteric methane emissions.

Additionally, the efficiency of the feed management system and the amount of feed wasted are important considerations for beef production LCAs. Using precision nutrition and feeding management can improve productivity by more accurately meeting individual or animal group nutrient requirements (Tylutki et al., 2008). Precision feeding of dairy cattle has been shown to decrease phosphorus excretion from dairy cattle and reduce costs (Cerosaletti et al., 2004; White and Capper, 2014). It is likely many feedlot operators have already implemented the principles of precision feeding on their operations due to tight operating margins; however, considering the variability in feeding management across operations may improve LCA estimates of uncertainty. Feed losses can occur during harvest, storage, and during the feeding

of the cattle, and while some partial beef LCAs have accounted for these losses (e.g. Stackhouse-Lawson et al., 2012; Rotz et al., 2013) most do not consider feed losses. As one of the principle benefits of ruminant agricultural systems is converting human inedible feedstuffs into human useable products, exploring the impacts of feed waste along the pre-farm gate beef supply chain with LCA methodology may provide insights into the influence efficiency has on the life cycle impacts of beef.

Animal health and welfare

Animal health and welfare play a vital role in the production of animal products. Public concern regarding the intensification of animal production systems has led to increased attention to animals in agriculture, extending the framework of animal welfare beyond health to include other considerations, such as the animal's ability to move and perform normal behaviors (Tucker et al., 2013). Additionally, as reviewed by Place and Mitloehner (2014), synergistic relationships likely exist between improving animal welfare and reducing environmental impacts, as animal health and welfare can affect animal productivity, economics, resource use, and waste outputs per unit of beef. Potential health and welfare issues that could be evaluated with LCA methodology include morbidity and mortality (particularly caused by bovine respiratory disease; BRD), transportation, and environmental stressors (e.g. heat stress).

Mortality in feedlots, while typically low, is a primary concern; however, disease may cost more than death loss due to treatment expenses, additional labor expenses, and reduced performance of cattle, with BRD being the most common disease of feedlot cattle (Smith, 1998). Snowden et al. (2006) evaluated the heritability, management implications, and economic effects of BRD in feedlot cattle and determined BRD-infected feedlot cattle had lower average daily gain and lower economic return compared to healthy cattle. Decreasing BRD incidence has the potential to reduce the environmental impact of feedlot systems through the reduction in resources required to produce the same amount of beef. In order to determine the effect of BRD in feedlot cattle on life cycle impacts on the environment (e.g. CO₂-eq emissions/kg of beef) it is necessary to evaluate the additional resources required for BRD-infected cattle to reach the market equivalent of healthy cattle.

Transportation is recognized as a common stressor in beef cattle production with animal welfare concerns expanding to include injury, fatigue, mortality and morbidity that may occur due to limited feed and water access, varying climatic conditions, noise and poor handling, and mixing with unfamiliar animals (Schwartzkopf-Genswein et al., 2012). Transportation and handling stress can lead to increased body weight loss and increased incidence of bruising and dark cutters (Arthington et al., 2003; Warren et al., 2010; Schwartzkopf-Genswein et al., 2012), which has implications for product yield and economics. Heat stress is estimated to cost the U.S. beef industry \$369 million annually, due to decreased performance, increased mortality, and decreased reproductive efficiency (St.-Pierre et al., 2003). However, the life-cycle impacts of heat stress on environmental impact and resource use per unit of beef have not been previously assessed. Conducting LCAs that account for the impacts of heat stress could provide insights into the nexus of beef cattle welfare, economics, and environmental impacts.

Food Waste

Currently, approximately 20% of edible beef is wasted in the United States (USDA, 2011). Losses at the consumer, retail, and foodservice portion of the beef supply chain mean all resource use, economic costs, and environmental impacts incurred earlier in the product chain have not contributed to human nutrition. Cuéllar and Webber (2010) concluded that the energy wasted in U.S. food waste (across all foods, not only beef) was greater than the amount of energy generated from the conversion of grains into ethanol. As most beef LCAs are not full cradle-to-grave (e.g. consumer) assessments, the impacts of food waste on the life-cycle impacts of beef production typically go unreported. Beyond simply reducing food waste, recycling food waste into waste streams other than landfills (e.g. anaerobic digestion) may be a promising solution to reducing environmental impacts. Considering the challenges

of feeding a growing world population with finite resources, reducing and recycling food waste may be an important strategy in sustainably meeting increased demand for animal protein.

Conclusion and Recommendations

Life cycle assessment is a powerful tool for evaluating the environmental impacts of producing beef. While most published LCAs focus solely on GHG emissions, expanding the impacts considered to other environmental emissions, such as reactive nitrogen, and economic and social concerns will likely improve the usefulness of LCAs. When comparing results across beef LCAs, consideration should be given to differences in geographic location, system boundaries, allocation methods, functional units, and the methodologies used to estimate environmental impacts. Ignoring the considerations above could lead to inappropriate comparisons and conclusions. Future LCAs of beef production should increase the use of process-based models that have been evaluated with experimental data when building input and output inventories. Expanding the use of process-based models is recommended because of the spatial and temporal variability of resource use and environmental emissions generated from beef production, due to the biogeochemical processes involved. Furthermore, increased consideration should be given in LCAs to aspects of the pre-farm gate supply chain that impact the production efficiency of beef, as there may be opportunities to improve economic, social (i.e. animal welfare), and environmental sustainability in concert. More full 'cradle-to-grave' LCAs of beef are needed to assess the impacts of food waste, and identify the potential impacts of reducing or recycling food waste. In conclusion, further strengthening the abilities of LCAs to capture the dynamic nature of the beef industry and harmonizing methods across LCAs will likely improve the accuracy of estimates and lead to more reporting of the uncertainties associated with results.

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For more information, contact:
Science and Product Solutions
National Cattlemen's Beef Association
A Contractor to the Beef Checkoff Program
9110 East Nichols Ave, Suite 300
Centennial, CO 80112
303.694.0305

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