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Uncertainties in life cycle greenhouse gas emissions from U.S. beef cattle

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Abstract

Beef cattle feedlots are estimated to contribute 26% of U.S. agricultural greenhouse gas (GHG) emissions, and future climate change policy could target reducing these emissions. Life cycle assessment (LCA) of GHG emissions from U.S. grain-fed beef cattle was conducted based on industry statistics and previous studies to identify the main sources of uncertainty in these estimations. Uncertainty associated with GHG emissions from indirect land use change, pasture soil emissions (e.g. soil carbon sequestration), enteric fermentation from cattle on pasture, and methane emissions from feedlot manure, respectively, contributed the most variability to life cycle GHG emissions from beef production. Feeding of coproducts from ethanol production was estimated to reduce life cycle emissions by 1.7%, but could increase emissions by 0.6–2.0% with higher feeding rates. Monte Carlo simulation found a range of life cycle emissions from 2.52 to 9.58 kg CO_2 per kg live weight (5th and 95th percentiles), with a calculated average of 8.14, which is between recent estimates. Current methods used by the U.S. Environmental Protection Agency (EPA) associated with beef production in feedlots were found to account for only 3–20% of life cycle GHG emissions.

Keywords: Beef cattle, Greenhouse gas emissions, Soil carbon, Land use change, Life cycle

1. Introduction

Climate change due to anthropogenic greenhouse gas (GHG) emissions is causing a range of environmental challenges and corresponding mitigation policies (Karl et al., 2009; Ellerman and Buchner, 2007). Global GHG emissions from livestock production were recently estimated to be 18% of anthropogenic GHGs (Steinfeld et al., 2006), which is roughly equal to emissions from all transportation systems globally, although this value has been disputed (Pitesky et al., 2009; Asner and Archer, 2010). Global livestock production accounts for 78% of agricultural land, 33% of all cropland for feed, and covers roughly 30% of terrestrial area (Steinfeld et al., 2006). Increasing population and rising living standards between 2000 and 2050 are expected to more than double global production of meat from 229 to 465 million metric tons (Steinfeld et al., 2006). In conjunction, by 2050, direct GHG emissions from meat, milk, and egg production are expected to increase by 39% above year 2000 levels (Pelletier and Tyedmers, 2010), yet many technologies could be developed or used to decrease these projected emissions levels. In the U.S. in 2010, direct GHG emissions from beef cattle totaled 112 Tg carbon dioxide-equivalent (CO_2e), corresponding to 26.2% of emissions from agriculture or 1.6% of national GHG emissions (EPA, 2012) (Table A.1 and A.2 in Supplementary Information).

Comprehensive and standardized methods to assess the environmental impact of livestock are now being developed by the Food and Agriculture Organization of the United Nations (FAO, 2013). Large agri-food companies and distributors (e.g. Kellogg's, BASF, Walmart, Bunge; http://www.Field-to-Market.org) have also recently begun initiatives to track the sustainability of agricultural products for potential marketing purposes (Fliegelman, 2010; NCBA, 2011). The full environmental impact of a product due to the extended impacts from its supply chain can be evaluated using life cycle assessment (LCA). Currently the National Cattleman's Beef Association is working with BASF on an LCA of beef production to identify the most important areas for future innovation to reduce environmental burdens.

In general, LCA has two uses: 1) to quantify absolute emissions from production life cycles for comparison with other systems, and 2) to identify system components to maximize efficiency and/or minimize environmental impacts. To quantify all GHG emissions from meat production from beef cattle, a comprehensive inventory of production inputs within a defined boundary is required, which includes impacts occurring away from facilities. Yet, a standard LCA boundary for GHG emissions from beef cattle is currently non-existent, which necessitates an investigation of possible significant emissions that may occur either directly or indirectly from production locations.

The multitude of related agricultural systems associated with beef production raises questions concerning appropriate system boundaries. The most extensive attributional LCA is only relevant to analyses with the same system boundaries, and could be less valuable for informing policy frameworks with a different set of boundaries, depending on the context and goals. Consequential LCAs that account for indirect effects from livestock production, such as related land use change that occurs nationally (Cederberg et al., 2011) and globally (Dumortier et al., 2012), provide a more expansive assessment of the real-world GHG emissions resulting from specific policies or actions. Because LCAs can have a range of complexity for the same system in question, particularly due to the inconsistent inclusion of indirect effects (Sanchez et al., 2012), all LCA results are relative to the specific system boundaries used and the analytical context. Comparison of these approaches can provide insight to the limitations of each quantification approach and can provide better guidance in designing new LCAs to guide innovation and to be used within policy frameworks where needed.

The life cycle GHG emissions intensity of U.S. beef cattle production has recently been estimated to range between 5.9 and 15.5 kg CO_2e per kg live weight (LW) (Hamerschlag, 2011; Pelletier et al., 2010; Phetteplace et al., 2001). Yet these studies did not include emissions from indirect land use change and use of co-products from biofuel production (Bellarby et al., 2013). Furthermore, these studies have insufficiently documented variability and uncertainty in these systems. To clarify these issues, an LCA of GHG emissions from beef production was conducted using U.S. industry statistics and previously published data to assess the contribution of land use and biofuel co-products, and an uncertainty analysis was constructed using the Monte Carlo method and a sensitivity analysis. The resulting LCA identifies factors in the life cycle of beef production that have the greatest sensitivity for determining emissions and thus require additional measurements to increase certainty in GHG emissions estimates. As environmental regulatory policy could employ LCA in the future, the LCA results were also compared with current regulatory policy frameworks.

2. Materials and Methods

2.1. System description

This study focuses on the cow-calf and feedlot systems of beef cattle production in the central United States. Consistent with standard LCA practice, output emissions are allocated by mass and are given on a per product basis (i.e. per unit live weight). System parameters were weighted by multiplying state-level data by the fraction of cattle in each state and summing the result (Table A.3 and A.4). Most beef calves (74%) are born between February and May while on pasture with the cow (EPA, 2010). At seven months, a designated number of beef heifers are chosen as "replacements" for breeding, while all steers and remaining heifers are fed on pasture (i.e. back-grounded) for ~0–17 months, depending on regional and temporal factors (such as availability of forage) and then transferred to feedlots.

2.2. Pasture

In addition to pasture and feedlot GHG emissions attributable to each slaughtered animal, each beef animal raised for meat is the product of a parent cow that lives on pasture. For this study, pasture GHG emissions were estimated using an arbitrary model cattle herd comprising 100 cows, 15 heifers, 3 bulls, and 90 spring-born calves (of which 75 are sent to finishing and 15 heifers kept as replacements) (Pelletier et al., 2010). Pasture GHG emissions of the entire herd were averaged over the yearly meat output of the herd, which included 75 feedlot-finished cattle (637 kg hd^{-1}) and 15 cull cows (636 kg hd^{-1}). Equations developed by the Intergovernmental Panel on Climate Change (IPCC) were used to estimate enteric fermentation methane emissions (EPA, 2010) (Table A.5). Literature estimates were used to evaluate the potential of pasture land to sequester carbon; however, much uncertainty and spatial variability exists in these estimates, which range from −3.5 to 1.2 kg CO_2e kg^{-1} LW (Pitesky et al., 2009; Asner and Archer, 2010) (Table A.6). For the resulting pasture portion of the LCA value, only emissions from enteric fermentation were counted; net soil GHG emissions and manure emissions were assumed to be zero, although variability was included in the uncertainty analysis. All pasture emissions are highly dependent on the number of cattle on pasture and the duration; this analysis does not include emissions from back-grounded cattle while on pasture, because aggregate statistics on this cattle population are not available. Several other GHG emissions including production of supplemental grain/forage, fossil fuel energy use for feeding, fuel for transportation of cattle, and embedded energy in equipment were excluded due to lack of data and their expected relatively smaller contribution compared to the other GHG emissions assessed (Liska et al., 2009).

2.3. Feed production for feedlots

Energy use emissions and other GHG emissions from corn production were attributed to the beef feedlot phase (Liska et al., 2009) (Table A.4). Additional emissions attributed to urea and forage in feed were calculated (Table A.7). For the LCA, GHG emissions from distiller's grains plus solubles (DGS) are assumed to be identical to corn grain by mass. Generally, some DGS are dried to facilitate storage and transportation (changes in emissions from energy to dry coproducts were not assessed); however, a scenario was modeled in which only wet DGS were fed at the maximum inclusion level (45%) to a subset of local feedlots near ethanol processing facilities (Max Wet DGS Use, Table 1).

2.4. Indirect land use change from feed grains and pasture

The approach used to estimate GHG emissions from land use change due to beef production allocates ongoing global land use change to aggregate global agricultural products (Steinfeld et al., 2006). A recent LCA of European beef and dairy cattle employed four different methods for estimating land use change (LUC) directly from the rate of grain consumption (Flysjö et al., 2011); however, some methods are not relevant to the U.S. since, unlike Europe, U.S. feed for beef cattle are not sourced from LUC sensitive areas (e.g. Brazil). The most general approach to LUC assumes "agricultural commodity markets are global and interconnected, and all demand for agricultural land contributes to commodity and land prices, and therefore contributes to land use change" (Audsley et al., 2009). This top-down method calculates the amount of LUC emissions attributable to global agriculture (assumed to be 58%) and divides it by global agricultural land regardless of use. Thus, for this study, it is assumed that GHG emissions of 1.43 Mg CO_2e are emitted due to LUC from each hectare of agricultural land used, including both crop and pasture (Audsley et al., 2009). Land use was determined by attributing all pasture acres to beef (subtract 2.8% of acres for lamb, USDA,

Table 1. Inventory of GHG emissions from US beef cattle feedlots (bold is emissions, italics are the main parameters that change in the scenarios).

Emissions sources	Units	Corn diet[a]	Current DGS use[a]	Max DGS use[a]	Max wet DGS use[a]
Pasture[b]					
Enteric fermentation (CH_4)[c]	kg CO_2e hd^{-1} yr^{-1}	2025	2025	2025	2025
	kg CO_2e kg^{-1} LW	**3.53**	**3.53**	**3.53**	**3.53**
Soil organic carbon sequestration[d]	kg CO_2e ha^{-1} yr^{-1}	**0**	**0**	**0**	**0**
Feed production (feedlot)					
Co-product inclusion level[e]	% DM intake	*0%*	*20%*	*45%*	*45%*
Average daily intake, coproduct[f]	kg hd^{-1} day^{-1}	*0*	*2.09*	*4.70*	*4.70*
Average daily intake, corn[g,h]	kg hd^{-1} day^{-1}	*9.14*	*7.05*	*4.44*	*4.44*
Days on feed[h]	Days	*192*	*177*	*185*	*177*
Urea[i]	kg CO_2e hd^{-1} yr^{-1}	71	–	–	–
Alfalfa hay[j]	kg CO_2e hd^{-1} yr^{-1}	4.3	4.3	4.3	4.3
Corn/DGS production[k]	kg CO_2e hd^{-1} yr^{-1}	914	914	914	914
	kg CO_2e hd^{-1} yr^{-1}	990	918	919	919
	kg CO_2e kg^{-1} LW	**0.89**	**0.76**	**0.80**	**0.85**
Land use change					
Land use change from pasture[m]	kg CO_2e kg^{-1} LW	**1.17**	**1.17**	**1.17**	**1.17**
Land use change from cropping[m]	kg CO_2e kg^{-1} LW	**0.85**	**0.85**	**0.85**	**0.85**
Feedlot					
Manure management[n]					
N-excretion rate	kg N hd^{-1} yr^{-1}	*69.5*	*82.2*	*103.8*	*103.8*
N_2O (direct & indirect)	kg CO_2e hd^{-1} yr^{-1}	733	865	1092	1092
CH_4	kg CO_2e hd^{-1} yr^{-1}	33	33	33	33
Enteric fermentation (CH_4)[n]	kg CO_2e hd^{-1} yr^{-1}	887	887	887	887
Feedlot fossil fuel use[p]	kg CO_2e hd^{-1} yr^{-1}	144	156	150	156
Soil organic carbon from manure[q]	kg CO_2e hd^{-1} yr^{-1}	0	0	0	0
	kg CO_2e hd^{-1} yr^{-1}	1797	1942	2162	2169
	kg CO_2e kg^{-1} LW	**1.62**	**1.61**	**1.88**	**1.80**
Emissions from processing[r]	kg CO_2e kg^{-1} LW	**0.07**	**0.07**	**0.07**	**0.07**
LCA total GHG emissions intensity	kg CO_2e kg^{-1} LW	**8.14**	**8.01**	**8.31**	**8.2**
Percent relative to conventional	%	100%	98.3%	102.0%	100.6%

a. Feed production scenarios: Conventional Corn Diet, Current use of distillers grains plus solubles (DGS), hypothetical Maximum DGS feeding rate (Bremer et al., 2010), and Max. Wet DGS Use used the same model, but was not previously published.
b. Several emission sources were not included; see methods.
c. Table A.5, kg CO_2e hd^{-1} yr^{-1} refer to a single cow on pasture; kg CO_2e kg^{-1} LW assumes 100 cows on pasture per 90 slaughtered animals (75 calves + 15 cull cows) (Pelletier et al., 2010); unit conversion assumes slaughter weight of 584 kg.
d. Soil carbon sequestration is highly variable and assumed to be zero for the baseline LCA, Table 2 and Table A.6.
e. Current DGS Use on average is composed of 24% dry DGS, 38% modified DGS, and 38% wet DGS; Maximum DGS Use is composed of 62% dry DGS, 19% modified DGS, and 19% wet DGS (Bremer et al., 2010).
f. f assume 87% of dry matter fed (10.45 kg hd^{-1} day^{-1}) (PCC, 2010) is corn or DGS; see "e" for percentages.
g. Corn subtract DGS.
h. (Bremer et al., 2010).
i. 1.22% dry matter, 0.127 kg hd^{-1} day^{-1}; 1.50 kg CO_2e per kg urea (Bremer et al., 2010).
j. 7.5% alfalfa in all scenarios, assume 114 kg CO_2 ha^{-1} GHG intensity, see Table A.7.
k. Corn and DGS are assumed to have same direct GHG intensity of 0.274 kg CO_2 kg^{-1} grain (Liska et al., 2009) Table A.4.
m. Land use change intensity for all agricultural land is 1.43 Mg ha^{-1} (Audsley et al., 2009), Table A.7.
n. Tables S3, S7, and S9.
p. Minnesota is characteristic of central U.S., Table 3.10, page 101 (Steinfeld et al., 2006).
q. (Schlesinger, 1999).
r. (Steinfeld et al., 2006).

2013) and land needed for feed production was estimated using corn yield (Bremer et al., 2010) (Table A.7). Unit conversions between kg CO_2e hd^{-1} to kg CO_2e kg^{-1} carcass weight assume 584 kg/hd, 192 days on feed, and 63% dressing percentage; conversion from kg CO_2e kg^{-1} carcass weight to kg CO_2e kg^{-1} beef; assume 75% meat:waste ratio.

2.5. Feedlot

Equations developed by the IPCC were used to estimate direct emissions of enteric methane as well methane and nitrous oxide (N_2O) from manure management (EPA, 2010), which is consistent with many other LCA studies (Crosson et al., 2011). In order to characterize U.S. dry feedlots (e.g. those without bedding or confinement), relevant industry data was used wherever possible in place of EPA or IPCC default parameter values. Proprietary data (including in weight, out weight, average weight, days on feed, average daily gain, and dry matter intake) from the Professional Cattle Consultants (PCC) published in monthly newsletters was compiled and analyzed (PCC, 2010) (Table A.3). Data from the PCC are defined by five U.S. cattle regions; in this study, data from the North Plains, Central Plains, and Corn Belt, comprising 11,575,000 steers and 9,635,000 heifers, are used to examine spatial variability (Figure A.1, Table A.8). Additional parameter values were used from the literature for feedlot performance, including energy content of feed dry matter (Vasconcelos and Galyean, 2007) and energy for maintenance (NRC, 2000) (Table A.9). Due to limited data, variables such as crude protein in the diet, fraction of gross energy converted to methane, and the ratio of net energy for maintenance to digestible energy were held constant throughout the analysis (Table A.4).

While several studies have shown that application of feedlot manure to nearby crop production systems increases soil carbon levels (Follett, 2001; Fronning et al., 2008), the net change in life cycle emissions from beef production was assumed to be zero since there exists a corresponding marginal decrease in soil carbon on the land used to produce

Table 2. Sensitivity analysis of uncertain factors in the life cycle for estimating GHG emissions from beef production.

Parameter	Units	Avg. ± SD *a*	High *b*	Low *c*	LCA GHG emissions, % change b*−c*/a*b*−c*/a*
Indirect land use change	kg CO_2e kg^{-1} LW	2.03[a]	2.03	−40.2[b]	500%
Pasture soil emissions	kg CO_2e ha^{-1} yr^{-1}	0	367[c]	−1102[d]	58.9%
Feedlot manure soil carbon	kg CO_2e hd^{-1} yr^{-1}	0[e]	−3088[f]	0[g]	34.1%
Enteric fermentation, pasture	kg CO_2e hd^{-1} yr^{-1}	2025[h]	2466[i]	1035[j]	16.0%
Manure methane emissions[k]	–	0.011[m]	0.44[n]	0.01	13.8%
Crop GHG intensity[p]	kg CO_2e Mg^{-1} grain	274	426	230	7.3%
Dry matter intake (feedlot)[q]	kg day^{-1}	8.8 ± 0.51	~9.8	~7.78	2.8%
Crop yield[r]	Mg ha^{-1}	9.56	10.7	7.22	2.2%
Animal mass (feedlot)[s]	kg	438.9 ± 26.2	~491.3	~386.5	0.5%
x^* = LCA output from use of x.					

a. First-derivative, allocation approach, sum of pasture and cropping, Table 1 (Audsley et al., 2009).
b. Second-derivative, consequential approach, maximum 85 kg CO_2e per kg beef reduced, dressing percentage of 63% and 75% meat:waste ratio for LW units (Dumortier et al., 2012).
c. 100 to −100 kg C ha^{-1} yr^{-1} change in soil organic C, low-input rangelands, Table 16.1 (Follett et al., 2001), Table A.6.
d. Flux from moderately grazed land, without enteric fermentation (Liebig et al., 2009), Table A.6.
e. Zero for LCA and Monte Carlo (Schlesinger, 1999).
f. (Fronning et al., 2008).
g. (Schlesinger, 1999).
h. From IPCC methods, Table 1.
i. Lactating cows emitting 391 L day^{-1} (McCaughey, 1999); trace gas methods consistently underestimate methane emissions because sampling measures only esophageal emissions.
j. (Chaves and Thompson, 2006).
k. Methane Conversion Factor (MCF) is most sensitive parameter within feedlot manure methane emissions. MCF = CH_4 generated/(volatile solids produces × max. CH_4 potential).
m. Table A.4.
n. Max MCF = 0.44 occurs in anaerobic conditions with cattle on deep litter, yields 13.8% change in GHG emissions. For Monte Carlo and LCA, dry feedlots (aerobic conditions) are assumed ranging from 0.01 to 0.015.
p. (Liska et al., 2009), Table A.4.
q. Dry matter intake is used to calculate Energy for Gain, key parameter for enteric methane emissions from the feedlot (PCC, 2010).
r. (USDA, 2013), Table A.4.
s. Animal mass impacts calculations for methane emissions from feedlot enteric fermentation (PCC, 2010).

grain/forage feed from which the manure carbon is derived (Schlesinger, 1999). The uncertainty of this parameter was quantified (Table 2), but it was not included in the Monte Carlo simulation because it was assumed to be zero based on Schlesinger (1999).

2.6. Uncertainty analysis and Monte Carlo simulation

Due to spatial and temporal factors that affect crop production and cattle performance, each component of the beef cattle life cycle has inherent variability. The relative impact of nine different parameters was assessed in a sensitivity analysis where high and low values were used in the LCA model and the relative change in emissions was calculated (Table 2). Monte Carlo simulation (i.e. stochastic iteration) was used to evaluate parameter variability and to generate a probability distribution of the expected LCA intensities. The program @Risk (Palisade Corporation, Ithaca, NY, http://www.palisade.com) was used to compute 10,000 iterations of outputs by varying seven parameters in a manner consistent with their probability of occurrence (Figure A.2); all parameters in the sensitivity analysis (Table 2), excluding ILUC and soil carbon from feedlot manure were incorporated. Pasture enteric fermentation and GHG emissions from pasture soils were assigned triangular distributions based on the extremes found in literature review (Chaves and Thompson, 2006; Follett et al., 2001; Liebig et al., 2009; McCaughey, 1999). Three other parameters (methane conversion factor, corn cropping GHG intensity, and corn yield) were assigned a discrete distribution characterized by frequencies determined by spatial weighting (Table A.4). Feedlot statistics such as dry matter intake and animal mass were assigned a normal distribution consistent with a known standard deviation.

2.7. EPA Methodologies

The EPA uses two methods for estimating GHG emissions. Annually since 1990, The Inventory of U.S. Greenhouse Gas Emissions and Sinks, referred to as "Annual Inventory," is consistent with methods developed by the IPCC for use within the Kyoto protocol (EPA, 2012 and IPCC et al., 2006). Alternatively, the Mandatory Reporting of Greenhouse Gases, referred to as "Mandatory Reporting," was created in 2009 to begin comprehensive data collection needed to inform future regulatory actions (Ellerman and Buchner, 2007; EPA, 2013a). The EPA requires U.S. GHG emitters across all industries to report emissions of more the 25,000 metric tons CO_2e per year. The 13,000 total facilities above this threshold encompass 85–90% of U.S. GHG emissions (http://ghgdata.epa.gov/ghgp/main.do).

Of the two EPA approaches, the Mandatory Reporting includes only emissions from manure management (Table A.10), while the Annual Inventory includes both manure management and enteric fermentation (Table A.9). The Annual Inventory method forms the basis for the pasture and feedlot sections of the LCA presented here; additionally, a feedlot-only scenario is presented (Figure 2) for comparison with the Mandatory Reporting output (which covers feedlot manure emissions only); equations used are shown in supplementary materials (Equations A.1 and A.2). Industry-derived parameters (ASABE, 2010; NRC, 2000; PCC, 2010; Vasconcelos and Galyean, 2007) were compared to EPA defaults (Tables A.9 and A.10). To characterize geospatial variability in feedlots, data used for animal mass, volatile solids, excreted nitrogen (N), ambient temperature, methane conversion factor, and fraction of N runoff/leaching were weighted into regional and national averages (EPA, 2009, 2013b) (Table A.3).

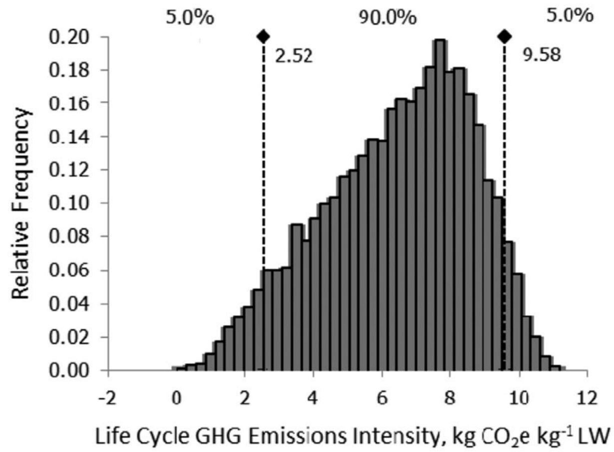

Figure 1. Probability distribution of life cycle GHG emissions from beef cattle using Monte Carlo simulation. Designations for 5th and 95th percentiles shown, based on parameters from Table 2, excluding land use change. See Tables S4 and Figure A.2.

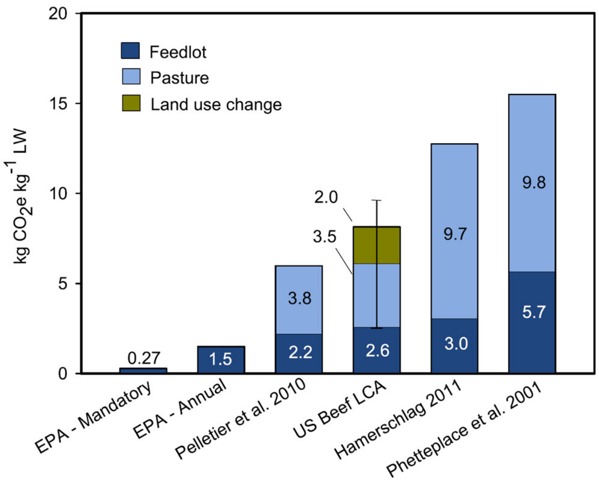

Figure 2. Comparison of estimated GHG emissions from the life cycle of US beef cattle production. US Beef LCA (Table 1), including error bars for max. and min. Monte Carlo simulation values (Figure 1) are compared to other LCA studies and EPA methods (Mandatory Reporting, Annual Inventory) (Table A.9 and A.10).

3. Results and discussion

3.1. Life cycle inventory

The average relative GHG emissions in the U.S. beef production life cycle were found to be from pasture (43%), indirect land use change (25%), feedlot (20%), and crop production for feed (11%). Aggregate GHG emissions sum to 8.14 kg CO_2e kg^{-1} LW (Table 1). Pasture emissions from the cow-calf system account for 3.53 kg CO_2e kg^{-1} LW of the beef life cycle, which is similarly estimated by other studies (3.76 kg CO_2e kg^{-1} LW) (Pelletier et al., 2010). Comparatively, Pelletier et al. reports that pasture GHG emissions occur primarily from enteric fermentation (~42%), feed production (~37%), and manure (~21%) and also assumes soil carbon sequestration at a rate of 0.12 kg C ha^{-1} yr^{-1} (Pelletier et al., 2010). Considerable uncertainty exists in the pasture component of the life cycle, especially with regard to stocking rate (Mu et al., 2013), length of backgrounding, and supplemental feeding.

As a result of the expansion of ethanol production from grain in recent years, use of coproducts as livestock feeds has increased (Bremer et al., 2010). Compared to corn grain, DGS has a higher energy density, which results in increased daily gain and less time in the feedlots (i.e. reduced days on feed) (Table 1). As DGS contain a larger fraction of protein than corn grain, N from urea is not added to DGS-supplemented feedlot diets. The higher N-content of DGS relative to corn correlates to an increase in excreted N and corresponding manure management N_2O emissions (Luebbe et al., 2012); it is currently not known how enteric fermentation is affected by DGS feeding. On average, DGS are fed at 20% of dry matter intake (Current DGS Use) when substituted in corn-based beef cattle diets (Corn Diet), which decreases feeding time by 8% and increases N-excreted by 18% (Table 1) (Luebbe et al., 2012). From a nutritional standpoint, inclusion of DGS in cattle diets is typically maximized at 45% of dry matter intake (Max DGS Use) (Bremer et al., 2010). At this increased feeding rate, a 4% reduction in time on feed and a 49% increase in N-excretion would result. Compared to the Corn Diet, Current DGS Use reduces emissions by 1.7% over the entire life cycle (Table 1). Emissions increase by 2.0% for Max DGS Use compared to Corn Diet, primarily due to increased N_2O emissions from greater N concentrations in manure. Feeding of only wet distillers grain at 45% inclusion (Max Wet DGS Use) is nearly equivalent to the Corn Diet, since the increased rate of gain when fed wet DGS, relative to modified or dry, further decreases time on feed (Table 1).

Within the feedlot portion of the life cycle, manure management and enteric fermentation contribute nearly equal emissions (766 and 888 kg CO_2e hd^{-1} yr^{-1}, respectively); additionally, direct emissions from fossil fuels used during feedlot production were included (144 kg CO_2e hd^{-1} yr^{-1}) (Table 1). Net emissions from soil carbon from manure application was assumed to be zero since there exists a corresponding marginal decrease in soil carbon on land used to produce grain/forage feed from which the manure carbon is derived (Schlesinger, 1999).

3.2. Indirect land use change from feed grains and pasture

Attributional LCA quantifies related production emissions that occur away from the feedlot facility but are caused by the supply chain of the system (see Materials and Methods). Alternatively, a consequential LCA accounts for changes in emissions from a range of sources that change as a consequence of production, and include emissions that are not part of the supply chain (Finnveden et al., 2009); emissions from indirect land use change (ILUC) are one of many possible consequential indirect emissions (Sanchez et al., 2012).

Emissions due to ILUC from production of corn-ethanol were recently estimated based on an increase in the mandated demand for ethanol (Searchinger et al., 2008). These ILUC emissions are currently included in regulatory LCAs for corn-ethanol at federal (EPA's Renewable Fuel Standard, RFS2) and state levels (California) (Hertel et al., 2010; Liska and Perrin, 2009; Searchinger et al., 2008). These studies use economic analysis to estimate the marginal change in grain price due to the change in grain demand, and the related change in global land conversion due to the increased grain price. Carbon dioxide emissions are then released from soils and standing biomass during deforestation from expansion of cropping areas. While livestock production clearly has a large role in land use trends, allocation of these emissions in an LCA requires careful consideration.

If the U.S. beef cattle population were increasing, such an additional GHG emission from ILUC from corn demand could be applied to the LCA, as calculated by the EPA for corn-ethanol. Alternatively, if the cattle population were decreasing, U.S. beef cattle would receive a GHG emission credit based on this ILUC calculation. The cattle population cycles due to various factors, and has generally been declining since the 1970's, but it is assumed for this analysis that the population is at a steady-state; in July 2006 and July 2011, beef cattle totaled 33.3 and 31.4 million, respectively (USDA, 2013). The EPA approach used for ILUC from corn-ethanol is additionally limited in its applicability to beef cattle because it considers corn production only (not pasture) and the economic modeling assumptions are predicated on a government-mandated increase in corn-ethanol production.

A recent examination of land use change explores potential future policy related to beef cattle (Dumortier et al., 2012). In that study, a 10% tax is imposed on U.S. steer prices. Modeled beef prices increase worldwide and global consumption decreases. Specifically, U.S. production decreases by 17.1%, however, this is countered by increased production in Argentina (4.8%), Brazil (4.9%), Canada (6.7%), Indonesia (4.0%), and elsewhere, since production in these areas is not subject to a tax. The resulting land use change in these countries is estimated to cause an increased emission of 37–85 kg CO_2e per kg of U.S. beef not produced. This result suggests that intensive U.S. beef production should be maintained in lieu of extensive production in carbon sensitive areas elsewhere (i.e. Brazil) in order to meet an inelastic global demand for beef. This consequential analysis is specific to the economic situation modeled (17% decrease in U.S. production) and is not appropriate for inclusion in an attributional LCA representing current average production.

The consequential ILUC emissions discussed above are calculated based on a change in the rate of production of a product; i.e. second derivative changes. Alternatively, consequential ILUC emissions can also be allocated to a product based only on the rate of production, and the associated rate of a related process; i.e. first derivative changes. The "first derivative" approach used here (see Materials and Methods) is useful in that it can be consistently applied to an allocational LCA framework in a transparent manner not dependent on external policies or short-term production trends. This ILUC methodology assumes all agricultural land contributes equally to ILUC, which simplifies calculations dramatically and has many advantages: 1) it allows allocation of ILUC emissions to all food types equally without the possibility of overcounting, 2) "food-types which have high land use requirements (e.g. beef) are allocated higher LUC emissions, and switching to food-types with lower land use requirements will show a reduction in LUC emissions," (Audsley et al., 2009) and 3) it attributes emissions to all commodities that utilize agricultural land regardless of arbitrary territorial boundaries. For U.S. beef production and this LCA, land use change emissions (LUC) are 0.85 and 1.17 kg CO_2e kg^{-1} LW for cropping and pasture land use, respectively (Table 1).

A drawback of Audsley et al.'s first-derivative approach is the assumption that some marginal pasture lands are currently only productive when used for cattle grazing (Foley et al., 2011), thus grazing lands are not contributing to LUC from the consequential perspective. But alternatively, recent analysis suggests grazing lands could also be used to power horse-based transportation (in the developing world) or for cellulosic biofuels, and thus have other uses and trade-offs (Liska and Heier, 2013); this would suggest pasture grazing should be attributed consequential ILUC emissions. Also, Audsley et al.'s approach does not capture increased soil carbon storage in pasture (compared to cropland) or address the impact of substitute products which may have an identical land area footprint, but different land use change impacts. Yet, this approach appears to be the most consistent with attributional LCA principles since it does not over-count and is not dependent on situational conditions. A second-derivative consequential framework, however, could be more insightful if needed to inform agricultural policy (Dumortier et al., 2012).

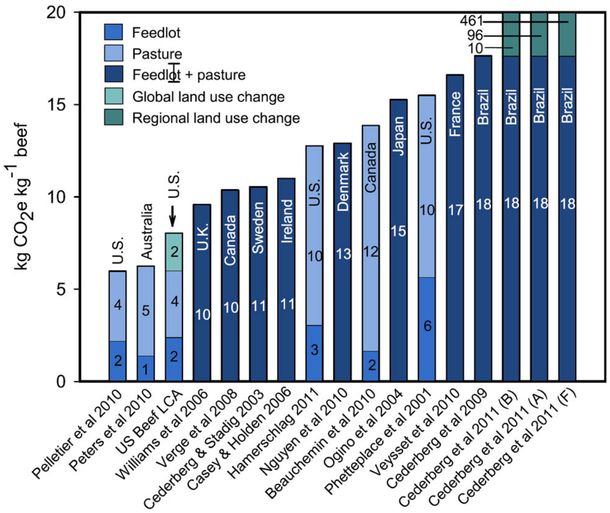

Figure 3. Comparison of previous global beef LCA estimates with the LCA presented here (kg CO_2e kg^{-1} LW), from Table 1, Table A.7 and A.9 (indicated with arrow); (B) Rainforest land use change averaged over all of Brazil; (A) Rainforest land use change averaged over the Legal Amazon Region; (F) Rainforest land use change averaged over newly deforested land.

3.3. *Uncertainty in the beef life cycle*

The sensitivity analysis performed here found that land use change clearly has the highest degree of uncertainty associated with beef production. Depending on the approach of the ILUC method (i.e. first-derivative vs. second-derivative), GHG emissions from land use change can vary 5-fold (Table 2). Alternate parameter values for pasture soil emissions, feedlot manure soil carbon, enteric fermentation of cattle on pasture, and manure management in the feedlot could change the LCA GHG intensity of beef by 14–59%. Other parameters had less of an effect (Table 2).

The Monte Carlo method generated a probability distribution of GHG intensities that range from 0.23 to 11.19 kg CO_2e kg^{-1} LW, with 90% of results between 2.52 and 9.58, and a standard deviation of 2.16 kg CO_2e kg^{-1} LW (Figure 1). The distribution is skewed to the left, due to the distribution of emissions from pasture that likely reduce the net GHG emissions, but (depending on pasture management) are possible to not sequester soil carbon in many situations (Follett et al., 2001) (Figure A.2). In the LCA (Table 1), net pasture soil carbon sequestration was assumed to be zero; inclusion of lower emissions due to sequestration in the Monte Carlo simulation was the biggest factor in shifting the calculated value of 8.14 kg CO_2e kg^{-1} LW to a Monte Carlo-derived mean of 6.46 kg CO_2e kg^{-1} LW. In general, quantifying these distributions of possible results is limited by lack of information concerning the distribution of most parameters; only seven were tested in this analysis. Thus, it is acknowledged that this approach does not fully capture the variability of the system.

3.4. *Comparison with other studies*

Estimates of the life cycle emissions intensity of beef production from previous studies were summarized and compared with this analysis (Figures 2 & 3). The life cycle emissions intensity reported here is very similar to a recent assessment for beef cattle in the U.S. (Pelletier et al., 2010), but the estimate provided here is roughly half of the intensity of two other estimates, at 12.8 and 15.5 kg CO_2e kg^{-1} LW (Hamerschlag, 2011; Phetteplace et al., 2001) (Figure 2). Previous estimates for U.S. beef were higher due to the unusual inclusion of mois-

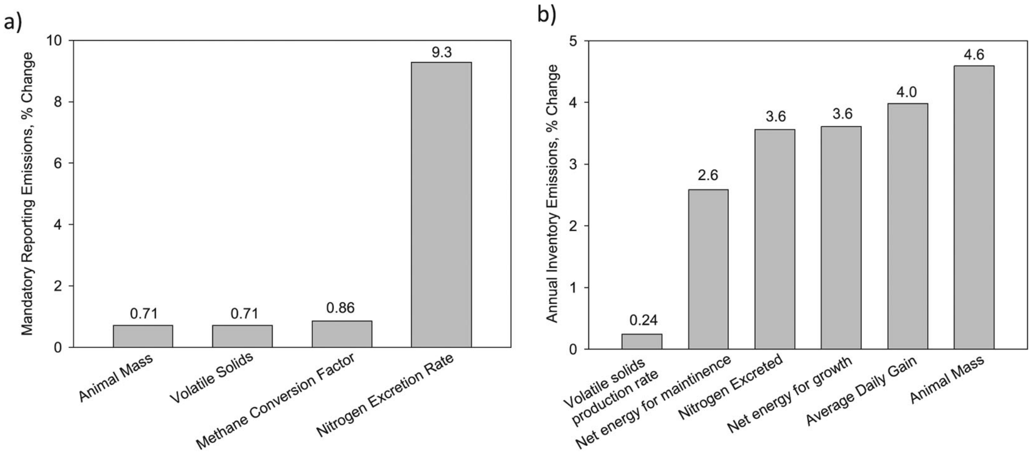

Figure 4. Sensitivity of parameters within EPA methodologies for quantification of cattle greenhouse gas emissions (kg CO_2e hd^{-1} yr^{-1}) with a ±10% change in the variable: a) Mandatory Reporting, b) Annual Inventory. From data in Tables A9 and A10 and additional calculations not shown.

ture and fat loss in cooking, plate loss, and spoilage (Hamerschlag, 2011). Other analysis had higher enteric fermentation emissions and higher ill-defined N_2O and pasture emissions (Phetteplace et al., 2001). Also, the LCA documented here excludes cattle during backgrounding, as no statistics are available on time and amount of cattle that are raised on pasture for a brief time prior to feedlot entry. The number of backgrounded cattle could be conservative, which could also explain the relatively low value found here (Tables A.5 and A.6). Adjusting the number of backgrounded calves to 70 for every 100 cows, a relatively high estimate, increases methane emissions from enteric fermentation by 34% to 4.73 kg CO_2e kg^{-1}. Life cycle emissions from other studies globally encompass Australia, North America, South America, Europe, and Asia and range from 5.9 kg CO_2e kg^{-1} LW in the U.S. (Pelletier et al., 2010) to 17.6 kg CO_2e kg^{-1} LW in Brazil (Cederberg et al., 2009) (Figure 3).

When considering LUC emissions, previous global assessments have indicated these emissions could range from 38 to 53% of all emissions from livestock (Asner and Archer, 2010; Steinfeld et al., 2006). A recent study in Brazil found deforestation associated with pasture expansion produced additional emissions in the range of 10.1–461 kg CO_2e kg^{-1} LW, depending on whether these regional land use change emissions are allocated to all beef cattle in Brazil, or to cattle only in areas of newly deforested land (Cederberg et al., 2011) (Figure 3).

3.5. Comparison of LCA with EPA Mandatory Reporting and Annual Inventory methods

Climate change mitigation policies exist at international, national, and state levels; however, they generally do not include livestock GHG emissions. The Kyoto protocol, following the international approach of the 1987 Montreal Protocol (ozone depleting chemicals) and the 1972 London Convention (marine waste dumping), is distinctive in accounting for livestock GHG emissions; however, the U.S. was not a participant (IPCC et al., 2006; Weiss and Jacobson, 1998). Carbon trading frameworks such as the European Union Emissions Trading Scheme (2005), the Regional Greenhouse Gas Initiative (2009, Northeastern States), and the AB 32 Global Warming Solutions Act (2012, California) do not quantify GHG emissions from livestock (Ellerman and Buchner, 2007). Livestock would have been eligible for financial support as "offsets" within the cap and trade system of the proposed *American Clean Energy and Security Act of 2009* (Waxman-Markey bill) but quantification procedures and metrics were never finalized (Rabe, 2010). In lieu of national legislative policy on climate change, a recent Supreme Court decision (*Massachusetts et al. vs. Environmental Protection Agency*) on April 2, 2007 specifically granted the EPA authority under the Clean Air Act to regulate GHG emissions (Mass v. EPA, 2007). In response, the Mandatory Reporting rule was created and a quantification framework for livestock was developed; however, the livestock section of the Mandatory Reporting rule was effectively eliminated by House Resolution 2996 in Section 425, which prohibits the EPA from using fiscal year 2010 appropriations to implement subpart JJ (Manure Management) of Part 98 of the Mandatory Reporting legislation (EPA, 2013b); this funding ban was further extended by the *Continuing Appropriations Act of 2011* (Public Law 111–242). While the government's role in climate issues is being debated, the majority of U.S. citizens believe climate change is real and support regulation by the government (Rabe, 2010).

Since there is clearly interest in GHG emissions from agricultural products from regulatory, political, and marketing perspectives, a comparison was made between the EPA feedlot quantification methods and other LCAs. Feedlots are the only phase of the production sequence where GHG emissions could be concentrated enough to meet the site-level regulatory threshold in the EPA's Mandatory Reporting methodology (25,000 Mg CO_2e per year), designated by the EPA as 29,300 head and encompassing an estimated 50 operations in the U.S. (EPA, 2009, 2013a). Feedlot GHG emissions estimated using the Mandatory Reporting methodology range from 300 kg CO_2e hd^{-1} yr^{-1} to 600 kg CO_2e hd^{-1} yr^{-1} (Table A.10) with GHG contributions being ~93% N_2O and ~7% CH_4. Variability in emission levels between three geographic regions was minimal, being less than 3–6% (Figure A.3). Sensitivity analysis shows that the N excretion rate has the greatest influence on final emissions (Figure 4a). Since this approach calculates only manure emissions from feedlots, these levels are significantly lower when converted to an emissions-per-product value (0.27 kg CO_2e kg^{-1} LW) and compared to an aggregate LCA (Figure 2).

The PCC industry statistics used with the EPA Annual Inventory methods formed the basis for a feedlot emissions estimate of 1653 kg CO_2e hd^{-1} yr^{-1}, ranging from 1590 to 1716 kg CO_2e hd^{-1} yr^{-1} (Table A.9). Spatial differences between the three PCC regions were 5% for EPA methods and industry values (Figure A.3). In a comparison of the sensitivity of five parameters (animal mass, daily gain, energy for growth, $N_{excreted}$, and energy for maintenance), variability of these factors had a roughly equal result on final values to be reported (Figure 4b). For industry data, the distribution of emissions was roughly ~55% for CH_4 from enteric fermentation, ~2% for CH_4 from manure management, ~39% from direct manure N_2O, and ~5% from indirect manure N_2O from runoff/leaching with subsequent volatilization. The EPA Annual Inventory forms the basis for direct feedlot emissions used in the LCA (1.49 kg CO_2e kg^{-1} LW) and is much lower relative to LCA values because it is only a component of the life cycle (Figure 2).

4. Conclusion

The U.S. *Energy Independence and Security Act of 2007* currently requires the use of LCA for quantifying GHG emissions from biofuels for comparison with petroleum products, and similar LCA methods are used in response to California climate policy (Bremer et al., 2010; Liska and Perrin, 2009; Liska et al., 2009). Thus, it is possible that future policy will use LCA to quantify emissions from other agricultural sectors. The quantitative methods underlying the EPA Mandatory Reporting rule are shown here to account for roughly 20% of feedlot GHG emissions (recognized by the more complete EPA Annual Inventory) and only 3% of life cycle GHG emissions (Figure 2). Yet, if an assessment of the total GHG emissions that result from beef cattle production were to be monitored or used for marketing purposes (Fliegelman, 2010; Hamerschlag, 2011), then a much higher emissions level, as documented here, would be expected compared to EPA's local assessments.

The uncertainty analysis performed here found that net pasture emissions and land use change have the highest degree of uncertainty associated with beef production. Longer pasture durations has been shown previously to have higher GHG emissions compared with shorter and more intensive feedlot finishing (Peters et al., 2010). Similarly, this analysis also found that the most GHG intense phase of the life cycle of beef cattle was during the pasture phase, and this was also associated with the greatest uncertainty in the life cycle. These results suggest that further research should focus on pasture level contributions to life cycle GHG emissions and validation of estimated feedlot emissions by direct measurement of GHG emissions.

The second-derivative and first-derivative approaches to ILUC from U.S. beef production produce dramatically different results in both sign and magnitude. Depending on the goals and scope of the LCA in question, either approach can be valid, but extensive effort should be made to clearly define the appropriateness of the application. In determining which approach is useful for government policy, first-derivative approaches could be used for national labeling programs, industry emissions thresholds, or comparison to other agriculture products, where appropriate. Alternatively, analysis of tax policies or other mandated management techniques should perhaps use a second-derivative methodology. Overall, it is clear that estimations of life cycle emissions from livestock are highly dependent on policy contexts, particularly concerning indirect effects, and results do not strictly reflect biophysical processes.

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Appendix A

Uncertainties in Life Cycle Greenhouse Gas Emissions from U.S. Beef Cattle

Quentin M. Dudley, Adam J. Liska, Andrea K. Watson, Galen E. Erickson

pages A1- A20

pg.

Table A.1 - Livestock emissions of CH_4 and N_2O in the United States ……………………………….…A1

Table A.2 - Fraction of U.S. emissions for agriculture from beef cattle. ……………………….…..…A2

Figure A.1 - Map of the Professional Cattle Consultants (PCC) regions in the central U.S.…..A2

Table A.3 - Spatial weighting of equation variables in the U.S.…………………………..…………..…A3

Table A.4 - Monte Carlo simulation input distributions using @Risk.… ……………….…..………A4

Figure A.2 - Probability distributions for input parameters into the Monte Carlo simulation using @Risk ……………………………………………………………………………..………………….………A5

Table A.5 - Methane emissions from enteric fermentation on pasture……………..………………A7

Table A.6 - Carbon sequestration in pastures……………………………………………..……….…….…A8

Table A.7 - Life cycle assessment components of beef cattle feedlot GHG emissions…….……A9

Table A.8 - Beef cattle regional performance data from the PCC…………………….……..……..A11

Table A.9 - EPA Annual Inventory of GHG emissions ……………………………………………..…..A12

Table A.10 - EPA Mandatory Reporting of GHG emissions..………………………………….………..A14

Figure A.3 - EPA methodologies (graphical results)…….……………………………………………..…A16

Equation A.1 - Annual Inventory equations……………………………………………….…………….A17

Equation A.2 - Mandatory Reporting equations...……………………………………………..……….…A18

References …………………………………………………………………………………………………….….….A20

Table A.1 - Livestock emissions of CH_4 and N_2O in the United States.

Gas/Animal type[a]	1990 Tg CO_2e	1990 % of total	2010 Tg CO_2e	2010 % of total
Methane from manure [b]				
Total U.S. livestock	31.7	100.0%	52.0	100.0%
Swine	13.1	41.3%	19.9	38.3%
Dairy Cattle	12.6	39.8%	26.0	50.0%
Poultry	2.8	8.8%	2.7	5.2%
Beef Cattle	*2.7*	*8.5%*	*2.8*	*5.4%*
Sheep	0.1	0.3%	0.1	0.2%
Horses	0.5	1.6%	0.5	0.9%
Nitrous oxide from manure [c]				
Total U.S. livestock	14.4	100.0%	18.3	100.0%
Beef Cattle	*6.3*	*43.8%*	*8.2*	*43.8%*
Dairy Cattle	5	34.7%	5.9	32.2%
Poultry	1.5	10.4%	1.6	8.7%
Swine	1.2	8.3%	1.9	10.4%
Horses	0.2	1.4%	0.3	1.6%
Sheep	0.1	0.7%	0.3	1.6%
Methane, enteric fermentation				
Total U.S. livestock	133.8	100.0%	140.6	100.0%
Beef Cattle	*96.2*	*71.9%*	*101.1*	*71.9%*
Dairy Cattle	31.8	23.8%	33.0	23.5%
Swine	1.7	1.3%	2.0	1.4%
Horses	1.9	1.4%	3.6	2.6%
Sheep	1.9	1.4%	0.9	0.6%

Source: Adapted from (EPA, 2012), Chapter 6, Table 6-3 and 6-6
[a] Totals may not sum due to independent rounding.
[b] Manure CH_4 includes emissions from anaerobic digestion
[c] Manure N_2O includes both direct and indirect emissions
http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2012-Main-Text.pdf

Table A.2 - Fraction of U.S. emissions for agriculture from beef cattle.

Gas/Source	U.S. Agr. 2010 Tg CO_2e	U.S. Beef Cattle in 2010 Tg CO_2e	Beef, %
Methane			
Total U.S. agricultural	202.2	103.9	51.4%
Enteric Fermentation	141.3	101.1	71.5%
Manure Management	52.0	2.8	5.4%
Rice Cultivation	8.6	-	-
Field Burning Ag. Residues	0.2	-	-
Nitrous oxide			
Total U.S. agricultural	226.2	8.2	3.6%
Agricultural Soils	207.8	-	-
Manure Management	*18.3*	*8.2*	*44.8%*
Field Burning of Ag. Residues	0.1	-	-
Total U.S. agricultural GHG	428.4	112.1	26.2%

	U.S. Total	Beef	Beef %
Total U.S. total GHG emissions	6,821.8	112.1	1.6%

Source: Adapted from (EPA, 2012), Executive Summary, Table ES-4; Chapter 6, Table 6-1, Table 6-3 and 6-6. http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2012-Main-Text.pdf

Figure A.1 - Map of the Professional Cattle Consultants (PCC) regions in the central U.S.

Source: http://www.pcc-online.com/

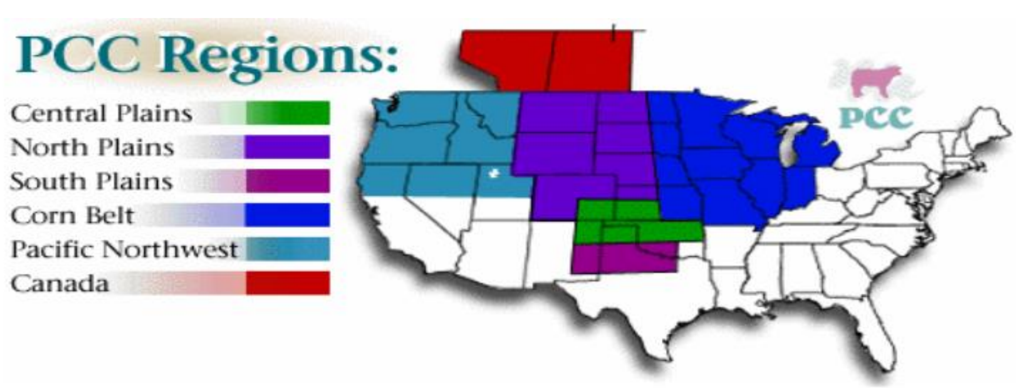


Table A.3 - Spatial weighting of equation variables in the U.S.

	% of state in PCC Region[a]	Cattle On feed[b]	Weighting Factor[c]	Mandatory Reporting (kg VS day^{-1} $1000kg^{-1}$)				Annual Inventory (kg $animal^{-1}$ $year^{-1}$)				Ambient Avg temp[d] (^{o}C)
				Volatile Solids		N excreted		Volatile Solids		N excreted		
				Steer	Heifer	Steer	Heifer	Steer	Heifer	Steer	Heifer	
Montana	63%	42,872	0.007	4.23	4.69	0.36	0.38	643.44	657.92	53.84	52.30	5.97
Wyoming	100%	79,567	0.022	4.17	4.61	0.35	0.37	654.09	671.24	54.83	53.46	5.54
Colorado	77%	1,130,652	0.240	3.97	4.34	0.33	0.35	665.37	685.65	55.87	54.70	7.30
North Dakota	64%	84,331	0.015	3.88	4.22	0.32	0.34	654.09	671.24	54.83	53.46	4.68
South Dakota	66%	17,783	0.093	4.01	4.39	0.34	0.35	656.55	674.32	55.05	53.73	7.30
Nebraska	63%	2,736,201	0.475	3.98	4.35	0.33	0.35	661.76	680.84	55.53	54.30	9.32
Kansas	20%	2,673,400	0.148	3.97	4.35	0.33	0.35	664.60	684.40	55.80	54.61	12.36
North Plains Average		-	1	3.98	4.36	0.33	0.35	662.14	681.39	55.57	54.34	8.92
North Dakota	36%	84,331	0.005	3.88	4.22	0.32	0.34	654.09	671.24	54.83	53.46	4.68
South Dakota	34%	517,783	0.030	4.01	4.39	0.34	0.35	656.55	674.32	55.05	53.73	7.30
Nebraska	37%	2,736,201	0.171	3.98	4.35	0.33	0.35	661.76	680.84	55.53	54.30	9.32
Kansas	52%	2,673,400	0.234	3.97	4.35	0.33	0.35	664.60	684.40	55.80	54.61	12.36
Minnesota	100%	610,752	0.103	3.89	4.42	0.33	0.34	669.49	690.51	56.25	55.15	5.09
Iowa	100%	1,738,545	0.294	3.93	4.28	0.33	0.34	657.78	675.86	55.17	53.87	8.78
Missouri	100%	83,007	0.014	4.08	4.49	0.34	0.36	662.08	681.24	55.56	54.34	12.47
Wisconsin	100%	277,759	0.047	3.95	4.31	0.33	0.34	658.08	676.24	55.19	53.90	6.18
Illinois	100%	311,976	0.053	4.15	4.59	0.35	0.37	648.76	664.58	54.33	52.88	10.97
Michigan	100%	179,158	0.030	4.00	4.38	0.34	0.35	656.99	674.88	55.09	53.78	6.89
Indiana	100%	105,264	0.018	3.98	4.35	0.33	0.35	646.10	661.25	54.09	52.59	10.91
Corn Belt Average		-	1	3.96	4.35	0.33	0.35	660.58	679.36	55.43	54.17	9.29
Colorado	23%	1,130,652	0.070	3.97	4.34	0.33	0.35	665.37	685.65	55.87	54.70	7.30
Kansas	28%	2,673,400	0.203	3.97	4.35	0.33	0.35	664.60	684.40	55.80	54.61	12.36
New Mexico	13%	154,556	0.006	3.88	4.22	0.32	0.33	651.18	667.61	54.56	53.15	11.91
Texas (3)	80%	3,056,260	0.660	3.95	4.32	0.33	0.34	660.25	678.95	55.40	54.14	18.24
Oklahoma	64%	357,906	0.062	3.98	4.35	0.33	0.35	655.46	672.96	54.95	53.61	13.83
Central Plains Average		-	1	3.96	4.33	0.33	0.34	661.14	680.09	55.48	54.24	15.98
Central US Average		13,250,744	-	3.97	4.35	0.33	0.35	661.17	680.12	55.48	54.24	-

[a] Region areas from Figure A.2 were analyzed with Image J software, ratio of pixels was compared
[b] USDA, 2013; use of "Quick Stats" database to find number of beef cattle "on feed" per state
[c] Texas, Percent of cattle in Central Plains is approximately 80% (based on approximation based on data from USDA, 2013); Avg temp is for Amarillo,TX to better represent northern region
[d] http://www.esrl.noaa.gov/psd/data/usclimate/tmp.state.19712000.climo

Table A.4 - Monte Carlo simulation input distributions using @Risk.

Normal Distributions

Parameter	Units	Average	Std. Dev.
Animal Mass[a]	kg	438.92	26.2
Daily Gain[a]	kg day^{-1}	1.37	0.11
Dry Matter Intake[a,b]	kg day^{-1}	8.80	0.51

Discrete Distributions

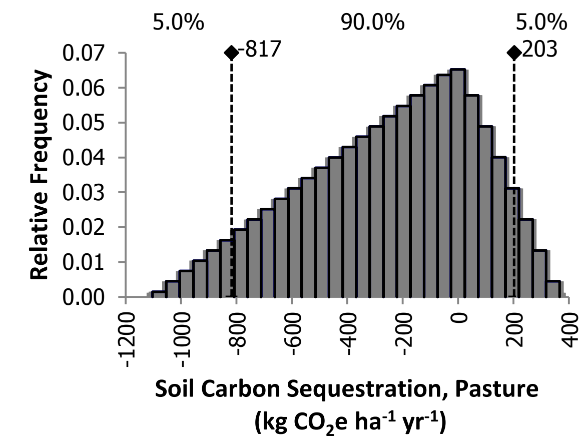
Methane Conversion Factor[d]	MCF value	Frequency
Cool (<14 ^{o}C)	0.015	0.2
Temperate (15-25 ^{o}C)	0.01	0.8
Weighted Average	0.0114	-

	Cropping Intensity[e] kg CO_2e Mg^{-1} grain	Grain Yield[e] Mg ha^{-1}	Frequency[f]
Colorado	316	8.72	0.0131
North Dakota	261	7.22	0.016
South Dakota	230	7.53	0.0544
Nebraska	301	9.73	0.1088
Kansas	327	8.47	0.0402
Minnesota	235	10.00	0.0935
Iowa	236	10.70	0.1675
Missouri	347	7.97	0.0394
Wisconsin	250	8.66	0.038
Illinois	274	10.20	0.1575
Michigan	290	8.47	0.0271
Indiana	287	9.79	0.076
Texas	426	7.78	0.0236
Ohio	311	9.54	0.0429
Kentucky	360	8.79	0.0155
Weighted Average	273.97	9.56	-

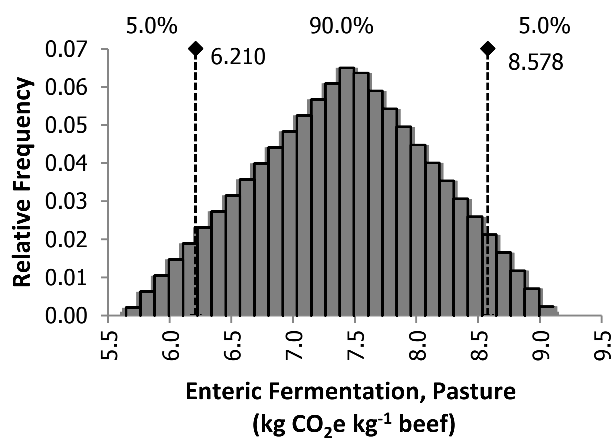
[a] See Table A.8
[b] Dry matter intake determines energy for gain parameter. See Table A.8[b]
[c] MCF
[d] Methane Conversion Factor (MCF), Table A-189, assume aerobic treatment and weighted average over central U.S. (Table A.5)(EPA, 2010). Frequency determined by state averages, see Table A.3
[e] USDA, 2013
[f] Determined by comparing levels of corn production for various states, average of years 2003-2005 (USDA, 2013)

Figure A.2 - Probability distributions for input parameters into the Monte Carlo simulation using @Risk. See Table 2 for references.

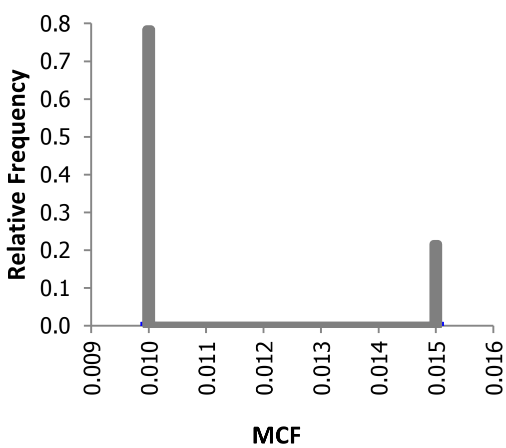
a) Soil Carbon Sequestration, Pasture



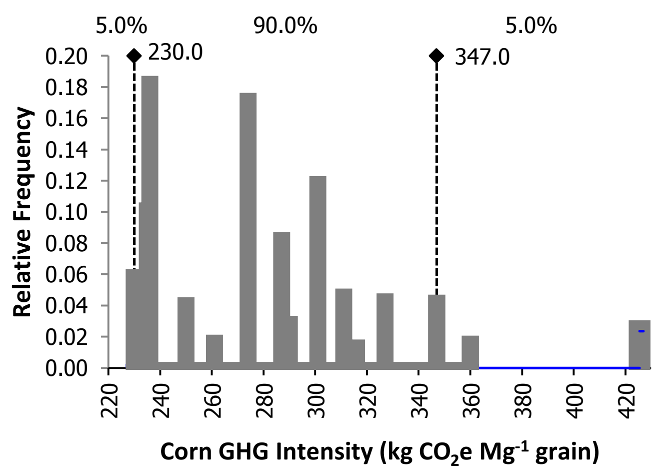
b) Enteric Fermentation, Pasture



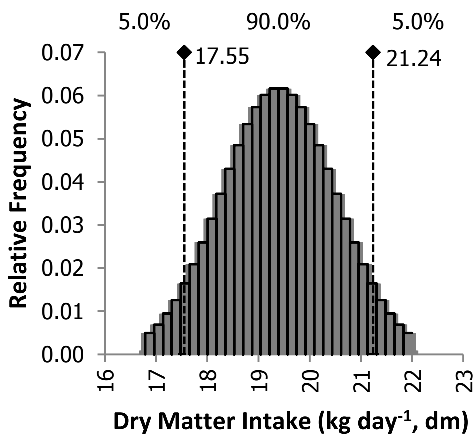
c) Manure Methane



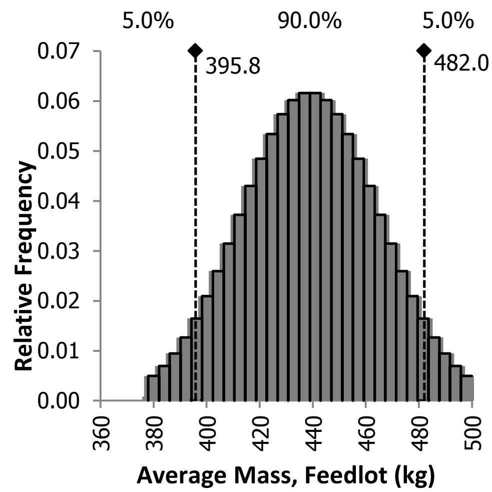
d) Crop GHG Intensity



e) Dry Matter Intake, Feedlot



f) Animal Mass, Feedlot



g) Crop Yield

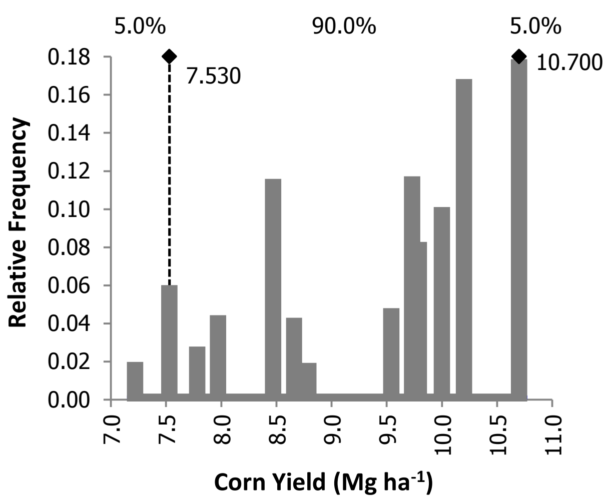


Table A.5 - Methane Emissions from Enteric Fermentation on Pasture, IPCC Methodology

Parameter/Emission Type	Units	
Methane, enteric fermentation		
Typical Animal Mass	kg	612.7[a]
Average Daily Gain	kg day^{-1}	0[b]
NE_m (Net energy for maintenance)	MJ day^{-1}	31.9[c]
NE_g (Net energy for growth)	MJ day^{-1}	0[b]
NE_a (Net energy for activity)	MJ day^{-1}	17.1[d]
NE_a (Net energy for lactation)[e]	MJ day^{-1}	-
NE_a (Net energy for pregnancy)[e]	MJ day^{-1}	-
REM (ratio of Nem to DE consumed)	-	0.510[f]
GE (Gross Energy)	MJ day^{-1}	198
CH_4 Emissions per cow on pasture	kg CO_2e hd^{-1} yr^{-1}	2025.2[g]
Life Cycle Impact of Enteric Emissions on Pasture		
Number of cows on pasture per 90 head slaughtered in feedlot	head	100[h]
Herd meat output	kg beef	27081[i]
Life Cycle Impact	kgCO_2e kg^{-1} beef	3.53

[a] Table A-171, assume beef cows and year 2009 (EPA, 2010)
[b] assume beef cattle who are not gaining
[c] assume CF_i =0.386. Chapter 10. Equation 10.3 and Table 10.4 (IPCC, 2006)
[d] assume CF_i =0.36 for large grazing areas. Chapter 10. Equation 10.4 and Table 10.5 (IPCC, 2006)
[e] Though mature female cattle have a lactation energy requirement for 7 months of the year and a pregnancy requirement for 9 months of the year, data on these energy demands are difficult to model and not standardized, thus they were not included in this analysis.
[f] Assume DE=64% (Table A-177, Northern Great Plains). Equation 10.14. Chapter 10. (IPCC, 2006)
[g] assume Y_m=6.5% (Table A-177, Northern Great Plains). DayEmit equation. Chapter 10. Page A-212. (IPCC, 2006)
[h] 100 cows on pasture per 90 head slaughtered is the sample herd assumed by Pelletier et al. (Pelletier et al., 2010), but could vary significantly as this value does not include backgrounded cattle. For example increasing this value to 135 (i.e. 70 of the 90 calves are backgrounded for ½ year), would increase methane emissions from enteric fermentation by 34% from 3.53 to 4.73 kgCO_2e kg^{-1} beef
[i] Assume 75 calves and 15 cull cows slaughtered each year for a sample herd (Pelletier et al., 2010). Assume slaughter weight of 636 kg
[j] Life cycle impact = CH_4 Emission per cow × Number of cows on pasture for sample herd (i.e. 100) / Sample herd meat output

Table A.6 - Carbon Sequestration in Pastures

Parameter/Emission Type	Units	
Pasture requirement for sample herd	Animal Units	159[a]
Stocking Density	AU $acre^{-1}$	0.35[b]
Herd pasture requirement[c]	ha	184
Herd meat output	kg beef	27081[d]
Land Intensity per unit beef	ha kg^{-1} beef	0.0068
Soil Carbon Sequestration Estimates		
High estimate for SOC sequestration[e]	$kgCO_2e\ ha^{-1}\ yr^{-1}$	-1102
Low estimate for SOC sequestration[f]	$kgCO_2e\ ha^{-1}\ yr^{-1}$	-166
Estimate for SOC emission[g]	$kgCO_2e\ ha^{-1}\ yr^{-1}$	367
Life Cycle Impact of SOC Estimates		
High estimate for SOC sequestration	$kgCO_2e\ kg^{-1}$ beef	-3.55
Low estimate for SOC sequestration	$kgCO_2e\ kg^{-1}$ beef	-0.53
Estimate for SOC emission	$kgCO_2e\ kg^{-1}$ beef	1.18

[a] Assume sample herd is 100 cows, 3 bulls, 15 replacement heifers, and 75 calves (Pelletier et al., 2010) at Animal Unit (AU) equivalents of 1, 1.35, 0.7,and 0.6 respectively. Note: no backgrounded calves are included.

[b] (Mu et al., 2012). Modification of this parameter (which has large spatial variability depending on seasonal fluctuations and weather patterns) would significantly impact output results

[c] Pasture requirement / Stocking Density

[d] Assume 75 calves and 15 cull cows slaughtered each year for a sample herd, as consistent with (Pelletier et al., 2010). Assume slaughter weight of 636 kg.

[e] moderately grazed pasture, flux value less enteric fermentation, Mandan, ND (Liebig et al., 2009)

[f] fertilized crested wheat grass, flux value less enteric fermentation, Mandan, ND (Liebig et al., 2009)

[g] 100 kg C $ha^{-1}yr^{-1}$ change in soil organic C for nonintensively managed rangelands Table 16.1. (Follett et al., 2001)

Table A.7 - Life cycle assessment components of beef cattle feedlot GHG emissions.

Parameter/Emission Type	Units	Conventional Corn Diet	Current DDG Use	Maximum DDG Use
Nitrogen Excretion and Manure				
$N_{excreted}$[a]	kg animal^{-1} yr^{-1}	69.65	82.2	103.8
N_2O Emissions (direct + indirect)	kgCO_2e hd^{-1} yr^{-1}	733	887	1107
Corn Feeding Rates				
Average Daily Intake[b]	kg hd^{-1} day^{-1}	10.45	10.45	10.45
Average Daily Intake (corn gain)[c]	kg hd^{-1} day^{-1}	9.14	7.05	4.44
Coproduct inclusion level[c]	% dry matter	0%	20%	45%
Daily Coproduct Intake	kg hd^{-1} day^{-1}	0	2.09	4.70
Urea Intake[c]	kg hd^{-1} day^{-1}	0.13	-	-
Urea Intensity[c]	kgCO_2e kg^{-1} urea	1.5	-	-
Urea Emissions	kgCO_2e hd^{-1} day^{-1}	0.195	-	-
Alfalfa Intake[c]	kg hd^{-1} day^{-1}	0.78	0.78	0.78
Alfalfa GHG Intensity[d]	kg C ha^{-1}	31.1	31.1	31.1
Average Yield[e]	short tons acre^{-1}	3.4	3.4	3.4
Alfalfa GHG Intensity	kgCO_2e kg^{-1} dm	0.015	0.015	0.015
Alfalfa GHG Emissions	kgCO_2e hd^{-1} day^{-1}	0.012	0.012	0.012
Direct Cropping GHG Emissions				
Corn Cropping GHG Intensity[f]	kgCO_2e kg^{-1} grain	0.274	0.274	0.274
Coproduct GHG Intensity[f]	kgCO_2e kg^{-1} grain	0.274	0.274	0.274
Feed GHG Emissions	kgCO_2e hd^{-1} day^{-1}	2.505	2.505	2.505
Land Use Change (LUC) GHG Emissions				
LUC Intensity, agricultural land[g]	kgCO_2 ha^{-1} yr^{-1}	1430	1430	1430
Corn Yield[h]	Mg ha^{-1} yr^{-1}	9.14	9.14	9.14
Corn or DGS Consumed[i]	kg hd^{-1} yr^{-1}	3337	3337	3337
Land Intensity of Corn[j]	ha hd^{-1}	0.349	0.349	0.349
Feed LUC Emission per head[k]	kgCO_2e hd^{-1} yr^{-1}	499.2	499.2	499.2
Total Cattle[m]	hd	32,834,801	32,834,801	32,834,801
Pasture Acres attributed to beef[n]	acres	38,801,937	38,801,937	38,801,937
Beef Pasture Density[p]	ha hd^{-1}	0.478	0.478	0.478
Pasture LUC Emission per head[k]	kgCO_2e hd^{-1} yr^{-1}	683.9	683.9	683.9
On-Farm Energy Use and GHG Emissions				
Diesel[q]	ton CO_2e hd^{-1}	0.047	0.047	0.047
LPG[q]	ton CO_2e hd^{-1}	0.015	0.015	0.015
Electricity[q]	ton CO_2e hd^{-1}	0.014	0.014	0.014
Total Energy GHG Intensity[q]	ton CO_2e hd^{-1}	0.076	0.076	0.076
Cattle on farm in one year[r]	cattle yr^{-1}	1.901	2.062	1.973
Fossil Fuel Use GHG Intensity	kgCO_2e hd^{-1} yr^{-1}	144.079	156.289	149.531

[a] assume $N_{excreted}$ increased by 18% (for 15% WDGS scenario) and 49% (for 30% WDGS scenario) which correlates to current DGS and the maximum DGS scenarios, respectively (Luebbe et al., 2012)
[b] (PCC, 2010)
[c] 0.784 kg hd^{-1} day^{-1} = 286 kg hd^{-1} yr^{-1} , values and scenarios from (Bremer et al., 2010)
[d] assume one seedling year, two established years, and one final year(Adler et al., 2007)
[e](USDA, 2013)
[f] assume spatial weighting over Central U.S region (Liska et al., 2009)
[g] (Audsley et al., 2009)
[h](Bremer et al., 2010) Central U.S. average, see Table A.3
[i] assume constant 9.14 kg hd^{-1} day^{-1} daily intake of corn or DGS
[j] corn consumed / corn yield
[k] Land intensity of corn (or pasture) * LUC Intensity of agricultural land
[m] (USDA, 2013) sum of state counts for cattle, cows, beef – inventory, 2007 Census of Agriculture
[n] derived from (USDA, 2013), sum of state counts for pastureland, 2007 Census of Agriculture (39,941,360 acres). Distribution between sheep and beef production was determined by economic value where "land attributed to beef" = total pasture acres * (109,900,000 lb beef slaughtered * $1.91/lb)/(109,900,000 lb beef slaughtered * $1.91/lb + 4,600,000 lb lamb & mutton slaughtered * $1.34/lb)
[p] pasture acres attributed to beef/ total cattle
[q] assume Minnesota is characteristic of central U.S., Table 3.10, page 101 (Steinfeld et al., 2006)
[r] inverse of days on feed from Table 1. (Bremer et al., 2010)

Table A.8 - Beef cattle regional performance data from the Professional Cattle Consultants.

		In Weight (kg)		**Out Weight (kg)**		**Average Weight (kg)**		**Days on Feed**		**Avg. Daily Gain (kg d^{-1})**		**Dry Matter Intake (kg d^{-1})**		**Energy for Gain (MJ d^{-1})[b]**	
		Value	**Std Dev**	**Value**	**Std Dev**	**Value**	**Std Dev[a]**	**Value**	**Std Dev**	**Value**	**Std Dev**	**Value**	**Std Dev**	**Value**	**Std Dev**
Central Plains	Heifers	312.69	10.94	530.34	12.73	421.52	11.87	164.28	11.55	1.32	0.07	8.48	0.36	21.79	2.26
	Steers	343.81	15.86	587.45	17.45	465.63	16.68	164.74	11.81	1.48	0.09	9.13	0.45	25.84	2.81
	Average	328.25	20.69	558.90	32.43	443.57	27.20	164.51	11.65	1.40	0.11	8.81	0.52	23.81	3.26
Corn Belt	Heifers	333.46	23.86	540.48	21.81	436.97	22.85	159.28	20.20	1.29	0.10	10.18	0.81	32.41	5.09
	Steers	356.02	28.23	598.13	26.02	477.08	27.15	162.84	18.68	1.47	0.11	10.68	0.87	35.60	5.47
	Average	344.74	28.41	569.31	37.53	457.02	33.29	161.06	19.49	1.38	0.14	10.43	0.88	34.00	5.51
North Plains	Heifers	345.40	24.17	558.34	26.05	451.87	25.13	158.89	19.94	1.34	0.10	9.31	0.70	26.96	4.40
	Steers	364.70	30.57	606.73	28.69	485.71	29.64	161.90	18.86	1.49	0.12	9.93	0.72	30.86	4.53
	Average	355.05	29.14	582.54	36.54	468.79	33.05	160.40	19.41	1.41	0.13	9.62	0.77	28.91	4.86
National Average	Heifers	308.84	11.16	528.32	13.21	418.58	12.23	170.22	10.41	1.29	0.07	8.51	0.38	21.96	2.37
	Steers	334.92	15.77	583.62	18.00	459.27	16.92	172.42	11.16	1.44	0.09	9.08	0.46	25.53	2.89
	Average	321.88	18.88	555.97	31.88	438.92	26.20	171.32	10.82	1.37	0.11	8.80	0.51	23.75	3.19

[a] Average Weight Standard Deviation = SQRT(($Stdev_{in}$*$Stdev_{in}$+$Stdev_{out}$*$Stdev_{out}$)/2)
[b] Energy for Gain = Total Dry Matter Intake multiplied by 1.5 Mcal kg^{-1} energy content (Vasconcelos and Galyean, 2007) less energy for maintenance (7.52 Mcal day^{-1}) (NRC, 2000)

Table A.9 - EPA Annual Inventory of GHG emissions.

Parameter/Emission Type	Units	EPA Assumptions Steer	EPA Assumptions Heifer	IPCC Assumptions Steer	IPCC Assumptions Heifer	Industry Assumptions Steer	Industry Assumptions Heifer	ASAE Data Avg.
Methane, enteric fermentation								
Typical Animal Mass	kg	457.7[a]	430.9	457.7[a]	430.9	459.3[b]	418.6	446.0[c]
Average Daily Gain	kg day^{-1}	1.41[d]	1.41	1.41[d]	1.41	1.44[b]	1.29	1.42[c]
NE_m (Net energy for maintenance)	MJ day^{-1}	31.9[e]	30.5	31.9[e]	30.5	31.5[f]	31.5	31.5[e]
NE_g (Net energy for growth)	MJ day^{-1}	30.0[g]	28.7	30.0[g]	28.7	25.5[h]	22.0	29.7[g]
GE (Gross Energy)	MJ day^{-1}	165.6[i]	158.3	165.6[i]	158.3	150.3[i]	139	163[i]
CH_4 Emissions[j]	kg CO_2e hd^{-1} yr^{-1}	1016.5	971.5	1016.5	971.5	922.7	852.3	1001.9
Methane from manure								
Volatile solids (VS)	kg $animal^{-1}$ yr^{-1}	661.2[k]	680.1	88.4[m]	84.5	547.5[n]	547.5	691.8[c]
CH_4 Emissions[p]	kg CO_2e hd^{-1} yr^{-1}	39.5	40.6	5.3	5.1	32.7	32.7	41.3
Nitrous oxide from manure								
$N_{excreted}$	kg $animal^{-1}$ yr^{-1}	55.5[q]	54.2	60.2[r]	56.6	69.7[s]	69.7	69.7[s]
Direct N_2O Emissions[t]	kg CO_2e hd^{-1} yr^{-1}	519.6	508.0	563.7	529.6	652.3	652.3	652.3
Indirect N_2O Emissions[u]	kg CO_2e hd^{-1} yr^{-1}	64.3	62.9	69.8	65.6	80.8	80.8	80.8
Total GHG emissions	kg CO_2e hd^{-1} yr^{-1}	1640.0	1583.0	1655.3	1571.7	1688.5	1618.1	1679.9
Total GHG emissions (average)	kg CO_2e hd^{-1} yr^{-1}	1611.5		1613.5		1653.3		
GHG Emissions for graphical comparison to LCA (see Figure 2, EPA - Annual)[v]	kg CO_2e kg^{-1} LW					1.49		

Note: the Annual Inventory methodology was developed to comply with international agreements and is based on IPCC methods (EPA assumptions column); for comparison, parameters such as volatile solids production rate and nitrogen excretion rate were calculated using the original IPCC equations for volatiles solids (VS) and $N_{excreted}$ parameters (IPCC assumptions column)

[a] Table A-171, assume feedlots and year 2009 (EPA, 2010)
[b] (PCC, 2010)
[c] (ASABE, 2010)
[d] Page A-206, 2.8 to 3.3 lbs day^{-1} (EPA, 2010)
[e] assume CF_i =0.322. Chapter 10. Equation 10.3 and Table 10.4 (IPCC, 2006)
[f] 450kg beef animal requires 7.52 Mcal day^{-1}. (31.46 MJ day^{-1} for maintenance) (NRC, 2000)
[g] Equation 10.6, assume body weight, castrates, mature body weight of female, and weight gain) (IPCC, 2006)

[h] assume $NE_m + NE_g$ = total energy intake. Average beef animal consumes 12.75 Mcal day^{-1} (Vasconcelos and Galyean, 2007). Subtract NE_m to get NE_g (Table A.8)
[i] Page A-212 in Section 3.9 (EPA, 2010)
[j] DayEmit equation. Page A-212 (EPA, 2010)
[k] Table A-186 (assume On Feed Beef Steer, Nebraska), cited from Moffroid and Pape, 2010 (EPA, 2010)
[m] see equation, Page A-216 (refers to IPCC2006 Tier II equations), assume UE=.02*GE for feedlot, assume ash content = .08 (EPA, 2010)
[n] assume 85% digestibility ("Beef Feed Nutrient Management Planning Economics (BFNMP$), version 2009. University of Nebraska-Lincoln. Lincoln, NE. http://water.unl.edu/web/manure/software#economics (accessed Jan 19, 2012)," 2009)
[p] Equation, Page A-222 (EPA, 2010)
[q] Total Kjeldahl N excretion rate, Table A-186 (assume On Feed Beef Steer, averaged over regions), cited from Moffroid and Pape, 2010 (EPA, 2010)
[r] Equations, Page A-217 (EPA, 2010), based on IPCC2006, Tier II equations and constants, assume percent crude protein = 13.34% (Vasconcelos and Galyean, 2007)
[s] based on 13.34% crude protein diet and 23 lb. intake; correlates to 27.48 kg N $animal^{-1}$ for 144 d feeding period changed to 365 d = 69.65 kg N $animal^{-1}$ yr^{-1} (Maximum value is potentially 98.12 based on 18% CP) (UNL, 2009)
[t] Equation, Page A-223 (EPA, 2010)
[u] Equation, Page A-224 (EPA, 2010)
[v] "EPA-Annual" as graphed in Figure 2 uses the steer-heifer average and industry assumptions. Conversion from kg CO_2e hd^{-1} yr^{-1} to kg CO_2e kg^{-1} LW assumes 192 days on feed, 584 kg slaughter weight.

Assumptions

Y_m (fraction of GE converted to CH_4) = 0.039, Table A-177, year 2009, steer/heifer feedlot (EPA, 2010)

Milk production, milk fat, and pregnancy all assumed to be 0

REM (ratio of NE_m to DE consumed) = 0.555, Equation 10.14 (IPCC, 2006)

REG (ratio of NE_g and DE consumed) = 0.375 Equation 10.15 (IPCC, 2006)

Standard Ref. Weight (mature female) = 500 kg

Net Energy for Activity (feedlot) = 0 MJ day^{-1} see page A-211, footnote #54 (EPA, 2010)

DE (% GE intake digestible) Table A-177, year 2009, steer/heifer feedlot (EPA, 2010)

Table A-187, assume dry lot

CH_4 production potential (B_o) = 0.33 m^3 CH_4 kg^{-1} VS (EPA, 2010). Table A-184 (assume Feedlot steers/heifers), cited from Hashimoto 1981 (EPA, 2010)

Methane Conversion Factor (MCF)=0.11, Table A-189, assume aerobic treatment and weighted average over central U.S. (Table A.3) (EPA, 2010)

Fraction of manure managed = 1, assume all manure is managed in feed lot

Direct N_2O emission factor (EF_{WMS}) = 0.02 kg N_2O kg^{-1} Kjeldahl N. Table A-191, assume dry lot (EPA, 2010)

$EF_{volitalization}$ = .010 kg N_2O-N/kg N. Indirect N_2O emission factor for volatilization, page A-224 (EPA, 2010)

$EF_{runoff/leach}$ = 0.008 kg N_2O-N/kg Indirect N_2O emission factor for runoff and leaching, page A-224 (EPA, 2010)

$Frac_{gas}$ = 23.0%. Fraction of N loss from volatilization of ammonia and NO_X, Table A-192, assume beef cattle on dry lot (EPA, 2010)

$Frac_{runoff/leach}$ = 2.35%. Fraction of N loss from runoff and leaching, Table A-192, assume beef cattle on dry lot and spatial average over central U.S. (EPA, 2010)

Table A.10 - EPA Mandatory Reporting of GHG emissions, includes manure only.

Parameter/Emission	Units	EPA Assumptions Steer	EPA Assumptions Heifer	Industry Assumptions Steers	Industry Assumptions Heifers	ASAE Data Avg.
Methane						
TAM_{AT}[a]	kg $head^{-1}$	420[a]	420	459.3[b]	418.6	446[c]
MCF_{MMSC} [d]	decimal	1.14%	1.14%	1.14%	1.14%	1.14%
VS_{AT}[e]	kg VS day^{-1} $1000kg^{-1}$	3.97[e]	4.35	1.58[f]	1.58	4.25[c]
Total CH_4 emissions[g]	kg CO_2e hd^{-1} $year^{-1}$	36.3	39.9	15.8	14.4	41.3
Nitrous oxide						
N_{AT}	kg VS day^{-1} $1000kg^{-1}$	0.33[e]	0.35	0.19[f]	0.19	0.37
Total Direct N_2O emissions[h]	kg CO_2e hd^{-1} $year^{-1}$	473.8	502.5	298.3	271.9	558.6
Total GHG emissions	kg CO_2e hd^{-1} $year^{-1}$	510.1	542.4	314.1	286.3	599.9
Total GHG emissions (average)	kg CO_2e hd^{-1} $year^{-1}$	526.3		300.2		
GHG Emissions for graphical comparison to LCA (see Figure 2, EPA - Mandatory)[i]	kg CO_2e kg^{-1} LW			0.27		

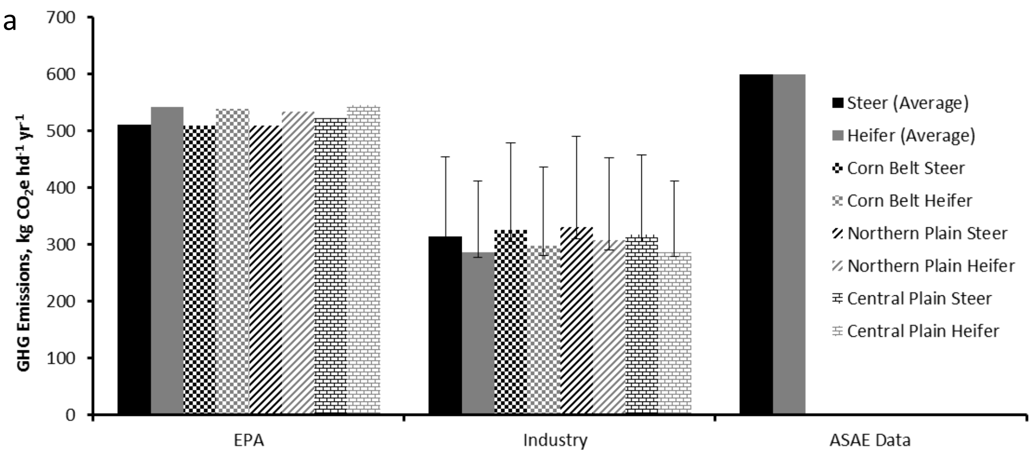
[a] TAM = typical animal mass, Table JJ-2 (EPA, 2009)
[b] (PCC, 2010)
[c] (ASABE, 2010)
[d] MCF = methane conversion factor (average), Table JJ-5, assume dry lots and average of 1.0% (cool ambient temp = <14 C°) and 0.5% (temperate ambient temp = 15-25 C°), weighted by number of cows in region. (EPA, 2009)
[e] VS = volatile solid excretion rate, N = nitrogen excreted per animal mass, Table JJ-2, assuming feedlot steers and spatial weighting of three-region average (EPA, 2009)
[f] 0.19 is average value, 0.269 is maximum value, (UNL, 2009)
[g] Equations JJ-2 and JJ-3. (EPA, 2009)
[h] Equations JJ-13 and JJ-14. (EPA, 2009)
[i] "EPA-Mandatory" as graphed in Figure 2 uses the steer-heifer average and industry assumptions. Conversion from kg CO_2e hd^{-1} yr^{-1} to kg CO_2e kg^{-1} LW assumes 192 days on feed, 584 kg slaughter weight.

Additional assumptions:
VS_{SS} (VS removal by solids separation) = 0 (no solid separation)
B_o (Maximum CH_4 conversion factor) = 0.33. Table JJ-2 (assume feedlot steers). (EPA, 2009)
VS_{MMSC} (fraction manure in system) = 1 (assume all manure is in dry lot feedlot)
EF_{MMSC} = 0.02 kgN_2O-N/kgN_2O. Table JJ-7 (assume drylot)

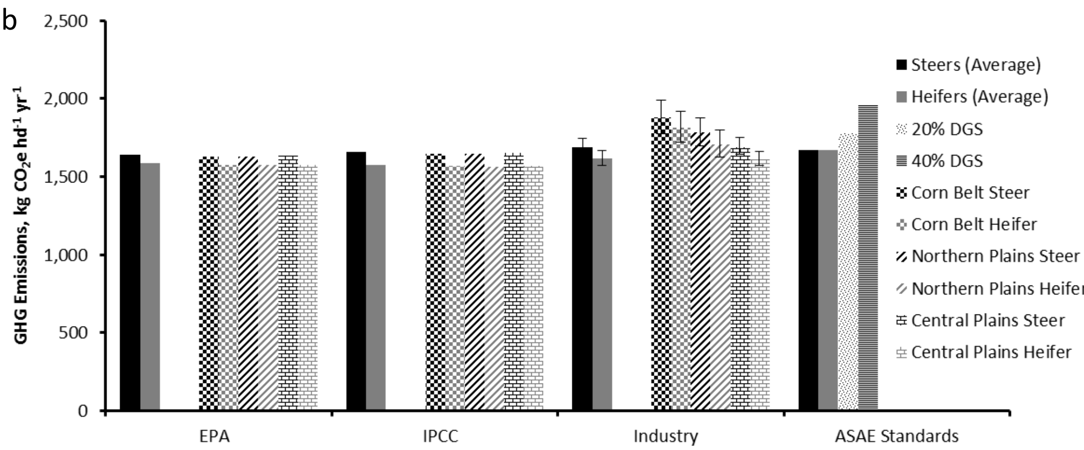
Comment on number of cattle to meet EPA threshold. To identify individual facilities for reporting GHG emissions, the EPA suggests that feedlots with a capacity of over 29,300 head would emit GHG emissions above the policy threshold of 25,000 Mg CO_2 yr^{-1} for Mandatory Reporting. Yet emissions for the 30,000 head feedlot discussed here total to 16,100 Mg CO_2e yr^{-1}, well under the emissions threshold. However, if maximum assumptions are used for volatile solids (5.25 kg VS day^{-1} $1000kg^{-1}$ animal mass), nitrogen excretion rate (0.42 kg VS day^{-1} 1000 kg^{-1} animal mass), and methane conversion factor (5%, solid manure storage), the threshold is nearly met (24,000 Mg CO_2e yr^{-1}). Deep bedding systems can also significantly increase emissions by utilizing a methane conversion factor (MCF) of 30% to 80%; for comparison, the drylot MCF used here is 1.5%. To minimize underreporting, it appears that the EPA's suggested reporting threshold of 29,000 head feedlot capacity that assumes the highest level of emissions per head.

Figure A.3 - EPA methodologies for quantification of cattle greenhouse gas emissions: a) Mandatory Reporting, b) Annual Inventory. From data in Tables A4 and A7, and additional calculations not shown.

a



b



Equation A.1 - Annual Inventory Equations (EPA, 2010)

Methane from Enteric Fermentation:

$$CH_4Emissions_{Enteric\ Fermentation}\ (\frac{kg\ CH_4}{head \times day}) = \frac{GE \times Y_m}{55.65}$$

Where $GE = Gross\ Energy = \left[\frac{\left(\frac{NE_m + NE_a + NE_l + NE_{work} + NE_p}{REM}\right) + \left(\frac{NE_g}{REG}\right)}{\frac{DE\%}{100}}\right]$

Where $Y_m = Fraction\ of\ GE\ converted\ to\ CH_4 = \ Ym(1990) \times \frac{e^{\frac{1.22}{(Year-1980)}}}{e^{\frac{1.22}{(1990-1980)}}}$

Methane from Manure Management:

$$CH_4Emissions_{Manure\ Management}\ \left(\frac{kg\ CH_4}{year}\right) = \ VS_{excreted} \times B_O \times MCF \times \frac{0.662\ kg\ CH_4}{m^3}$$

Nitrous Oxide from Manure Management:

$$Direct\ N_2O\ Emissions = \ N_{excreted}\ \times EF_{WMS} \times \frac{44}{28}$$

$$\begin{aligned} Indirect\ N_2O\ Emissions \\ &= \left(\ N_{excreted} \times \frac{Frac_{gas,WMS}}{100} \times EF_{volatilization} \times \frac{44}{28}\right) \\ &+ \left(\ N_{excreted} \times \frac{Frac_{runoff,leach\ ,WMS}}{100} \times EF_{runoff,leach} \times \frac{44}{28}\right) \end{aligned}$$

Where (EPA default method): $N_{excreted} = Population\ \times WMS \times N_{Ex}$

Where (IPCC default method):

$$N_{excreted} = N_{consumed} - (N_{growth} + N_{milk}) = \left(\frac{GE}{18.45} \times \frac{\frac{CP\%}{100}}{6.25}\right) - \left(\frac{\frac{\{WG \times [268 - \frac{7.03 \times NEg}{WG}}{1000}}{6.25}\right)$$

Equation A.2 - Mandatory Reporting Equations (EPA, 2009)

Methane from Manure Management:

$$Equation\ JJ-2 = CH_4Emissions_{Manure\ Management\ Systems}\ (\frac{metric\ tons}{year})$$

$$= \sum \left[(TVS_{AT} \times VS_{MMSC} \times (1 - VS_{SS}) \times \frac{365\ days}{1\ year} \times \frac{0.33\ m^3 CH_4}{kg\ VS\ added} \times MCF_{MMSC}) \times \frac{0.662\ kg\ CH_4}{m^3} \times \frac{1\ metric\ ton}{1000kg} \right]$$

Where $TVS_{AT} = Total\ Volatile\ Solids_{Animal\ Type}\ (\frac{kg}{day}) = Population \times \frac{420\ kg}{head} \times VS_{AT}/1000$
Where $VS_{MMSC} = Fraction\ of\ total\ manure\ managed\ in\ the\ manure\ system$
Where $VS_{SS} = Volatile\ solids\ removed\ through\ solid\ separation$

Direct Nitrous Oxide from Manure Management:

$$Equation\ JJ-13 = Direct\ N_2O\ Emissions\ (\frac{metric\ tons}{year})$$

$$= \sum \left[(N_{ex,AT} \times NVS_{ex,MMSC} \times (1 - N_{SS}) \times EF_{MMMSC} \times \frac{365\ days}{1\ year}\ ) \times \frac{44\ N_2O}{28\ N_2O-N} \times \frac{1\ metric\ ton}{1000kg} \right]$$

Where $N_{ex,AT} = Total\ Nitrogren\ Excreted_{Animal\ Type}\ (\frac{kg}{day}) = Population \times \frac{420\ kg}{head} \times \frac{N_{AT}}{1000}$

Total Emissions from Mandatory Reporting Methodology:

$$Equation\ JJ-15 = Total\ Emissions(\frac{metric\ tons\ CO_2e}{year})$$

$$= [(CH_4emissions_{MMS} + CH_4emissions_{AD}) \times 21] + [Direct\ N_2O\ emissions \times 310]$$

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