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Life Cycle Boundaries and Greenhouse Gas Emissions from Beef Cattle

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LIFE CYCLE BOUNDARIES AND GREENHOUSE GAS EMISSIONS FROM BEEF CATTLE

by

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Beef cattle are estimated to directly contribute 26% of U.S. agricultural greenhouse gas (GHG) emissions, and future climate change policy may target reducing these emissions. Life cycle assessment (LCA) of GHG emissions from U.S. feedlot beef cattle was conducted to compare methods of the U.S. Environmental Protection Agency (EPA) with a more complete evaluation of emissions. The inclusion of emissions from crop production for feed, associated land use change, and other minor factors nearly doubled GHG emissions associated with beef feedlots from the EPA Annual Inventory estimate of 1611 kgCO_2e hd^{-1} yr^{-1} to 3182 ± 167 kgCO_2e hd^{-1} yr^{-1}. Feeding of coproducts from ethanol production is estimated to reduce feedlot emissions by 6%. Furthermore, inclusion of pasture and land use change emissions from the cow-calf stage of the animal life cycle nearly tripled GHG emissions compared to the feedlot LCA (6.0 to 16.67 ± 0.32 kgCO_2e kg^{-1} beef). Despite use of expanded system boundaries in the LCA, U.S. beef cattle GHG emissions were lower than the majority of previous U.S. and international assessments of beef cattle. Nearly a 16-fold range in results can be found for U.S. beef using different system boundaries and assumptions. Use of LCA-driven carbon pricing on U.S. beef could reduce beef demand and associated beef GHG emissions by 2.7 to 21 Tg CO_2e yr^{-1}.

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Chapter 1: Climate Change Policy and Livestock

1.1 Greenhouse Gas Emissions from Livestock

Climate change due to anthropogenic greenhouse gas (GHG) emissions is leading to a range of environmental challenges and corresponding mitigation policies (Karl, Melillo, & Peterseon 2009). Global GHG emissions from livestock production were recently estimated to be 18% of total anthropogenic GHGs (Steinfeld, Gerber, Wassenaar, Castel, & de Haan, 2006), which is roughly equal to emissions from all transportation systems globally, although this value has been disputed (Asner & Archer, 2010; Pitesky, Stackhouse, & Mitloehner, 2009). Global livestock production accounts for 78% of agricultural land, 33% of all crop land for feed, and covers roughly 30% of terrestrial area (Steinfeld et al., 2006). Increasing population and rising living standards between 2000 and 2050 are projected to more than double the demand for meat from 229 million tonnes per year to 465 million tonnes per year (Steinfeld et al., 2006). In conjunction, by 2050, direct GHG emissions from meat, milk, and egg production are projected to increase by 39% above year 2000 levels (Pelletier & Tyedmers, 2010), yet many technologies may be developed or used to decrease these projected emissions levels.

Although beef cattle are estimated to be a relatively large source of GHG emissions, accurate quantification of these GHG has considerable uncertainty due to the use of inconsistent system boundaries, timeframes, and metrics (Crosson et al., 2011).

The Intergovernmental Panel on Climate Change (IPCC) has defined a methodology for estimation and monitoring of national GHG emissions, and these methods form the basis for U.S. Environmental Protection Agency (EPA) calculations of livestock emissions (IPCC, 2006). Direct emissions from beef cattle production are estimated by the EPA based on three localized sources: methane from enteric fermentation, methane from manure decomposition, and nitrous oxide from manure (both direct and indirect) (EPA, 2010a). Cattle also emit signficant amounts of carbon dioxide via respiration and enteric fermentation, but these emission sources are not counted by GHG assessment methodologies since this carbon originated in feed that was captured from the atmosphere via photosynthesis; cattle respiration merely returns this carbon to the atmosphere.

According to the EPA, GHG emissions from beef cattle in the United States totaled 110.7 Tg carbon dioxide equivalent (CO_2e) in 2008, equivalent to 25.9% of emissions from agriculture (EPA, 2010a). These emissions are approximately 1.6% of total GHG emissions in the U.S., which is somewhat larger than emissions from the U.S. military (Liska & Perrin, 2010). About 72% of all U.S. enteric methane emissions come from microbial digestion in the rumen of beef cattle (EPA, 2010a), and manure from beef cattle contributes about 43% of manure N_2O emissions from all U.S. livestock (Table 1; Table 2).

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

Gas/Animal type[a]	1990 Tg CO_2 Eq.	1990 % of total	2008 Tg CO_2 Eq.	2008 % of total
Methane from manure [b]				
Total U.S. livestock	29.3	100.0%	45.0	100.0%
Swine	13.1	44.7%	19.6	43.6%
Dairy Cattle	10.2	34.8%	19.4	43.1%
Poultry	2.8	9.6%	2.6	5.8%
Beef Cattle	*2.6*	*8.9%*	*2.5*	*5.6%*
Sheep	0.1	0.3%	0.8	1.8%
Horses	0.5	1.7%	0.1	0.2%
Nitrous oxide from manure [c]				
Total U.S. livestock	14.4	100.0%	17.1	100.0%
Beef Cattle	*6.3*	*43.8%*	*7.4*	*43.3%*
Dairy Cattle	5	34.7%	5.5	32.2%
Poultry	1.5	10.4%	1.8	10.5%
Swine	1.2	8.3%	1.7	9.9%
Horses	0.2	1.4%	0.4	2.3%
Sheep	0.1	0.7%	0.3	1.8%
Methane, enteric fermentation				
Total U.S. livestock	132.0	100.0%	140.6	100.0%
Beef Cattle	*94.5*	*71.6%*	*100.8*	*71.7%*
Dairy Cattle	32.0	24.2%	33.1	23.5%
Swine	1.7	1.4%	2.1	1.5%
Horses	1.9	1.4%	3.6	2.6%
Sheep	1.9	0.0%	1	0.7%

Source: Adapted from EPA (2010a). 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

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

Gas/Source	U.S. Agr. 2008 Tg CO_2 Eq.	U.S. Beef Cattle in 2008 Tg CO_2 Eq.	 Beef, %
Methane			
Total U.S. agricultural	194.0	103.3	77.3%
Enteric Fermentation	140.8	100.8	71.7%
Manure Management	45.0	2.5	5.6%
Rice Cultivation	7.2	-	-
Field Burning Ag. Residues	1.0	-	-
Nitrous oxide			
Total U.S. agricultural	233.5	7.4	3.2%
Agricultural Soils	215.9	-	-
Manure Management	*17.1*	*7.4*	*43.3%*
Field Burning of Ag. Residues	0.5	-	-
Total U.S. agricultural GHG	427.5	110.7	25.9%

	U.S. Total	Beef	Beef %
Total U.S. total GHG emissions	6,956.8	110.7	1.6%

Source: Adapted from EPA (2010a). Executive Summary, Table ES-4; Chapter 6, Table 6-3 & 6-6.

1.2 Existing Climate Change Policies

Climate change mitigation policies generally do not include livestock GHG emissions. The Kyoto protocol is the only climate policy that accounts for livestock GHG emissions at this time, and the U.S. is not a participant (IPCC, 2006). Livestock are not included in the largest active cap-and-trade system globally, the European Union Emissions Trading Scheme which began in 2005 (Ellerman & Buchner, 2007). Other regional cap-and-trade systems within the United States, specifically the Regional Greenhouse Gas Initiative in the Northeastern states and the Global Warming Solutions Act (AB 32) of California, also do not account for GHG emissions from livestock. The Netherlands imposes some of the most extensive air quality requirements for livestock (including odor, particulate matter, SO_x, NO_x, volatile organic compounds, and ammonia), but do not specifically include GHG emissions at this time (Melse, Ogink, & Rulkens, 2009).

Historically, international treaties have been precedents for environmental regulatory policies at the national level. For example, the 1987 Montreal Protocol and the 1972 London Convention were both responsible for defining U.S. emission levels concerning ozone depleting chemicals (e.g. chlorofluorocarbons) and marine waste dumping, respectively (Weiss & Jacobson, 1998). In both cases, some U.S. legislation preceded international consensus, but once international treaties were in place, U.S. law conformed to a more restrictive international standard. The U.S. has yet to formally participate in international climate change agreements, but 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 (Massachusetts v. EPA, 2007). Indicating the direction of related policy developments, the proposed *American Clean Energy and Security Act of 2009* (Waxman-Markey bill) outlined a national cap-and-trade system and was passed by the U.S. House of Representatives, but was not approved by the Senate.

Based on a U.S. national surveys in 2008 and 2009, “most U.S. citizens believe not only that global warming exists but also that it is a serious problem facing the nation…[and] most Americans believe that immediate government action is needed to deal with climate change and that governments at all levels of the federal system have a responsibility" (Borick, 2010, p. 54). In sum, momentum for climate mitigation and potential legislation appears to be growing; thus, discussion concerning the accuracy and scope of relevant quantification methodologies is needed to inform future policy.

1.3 Current EPA Quantification Methodologies

The EPA uses two methods for estimating GHG emissions. The Inventory of U.S. Greenhouse Gas Emissions and Sinks has been calculated annually since 1990 and is consistent with the IPCC methods used by the Kyoto protocol (EPA, 2010a). In response to the 2007 Supreme Court ruling, the Mandatory Reporting of Greenhouse Gases was created in 2009 under the U.S. Consolidated Appropriations Act of 2008 to begin comprehensive collection of data needed to inform future regulatory actions (Ellerman & Buchner, 2007; EPA, 2012a). The rule requires U.S. GHG emitters across all industries to report emissions of more the 25,000 metric tons carbon dioxide equivalent (CO_2e) per year. The 13,000 total facilities above this threshold encompass 85-90% of U.S. GHG emissions, and results of the national survey are now publically available (http://ghgdata.epa.gov/ghgp/main.do). 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, 2012b). This funding ban was further extended by the Continuing Appropriations Act of 2011 (Public Law 111-242).

1.4 Life Cycle Assessment

In response to recent trends in climate change policy, there is a need for a more complete and accurate understanding of GHG emissions from beef cattle. Life cycle assessment (LCA) is a method to determine the full environmental impact of a product due to the extended impacts from its supply chain, and it can be used to comprehensively estimate GHG emissions. The U.S. Energy Independence and Security Act of 2007 currently uses LCA for quantifying GHG emissions from biofuels, and similar LCA methods are used in response to California climate policy (Bremer et al., 2010; Liska et al., 2009; Liska & Perrin, 2009). With varying results, LCA has recently been applied to determine the total GHG emissions from intensive beef cattle production (Beauchemin, Henry Janzen, Little, McAllister, & McGinn, 2010; Casey & Holden, 2006; Cederberg, Meyer, & Flysjö, 2009; Cederberg, Persson, Neovius, Molander, & Clift, 2011; Cederberg & Stadig, 2003; Crosson et al., 2011; de Vries & de Boer, 2010; Gurian-Sherman, 2011; Hamerschlag, 2011; Nguyen, Hermansen, & Mogensen, 2010; Ogino, Kaku, Osada, & Shimada, 2004; Pelletier, Pirog, & Rasmussen, 2010; Peters et al., 2010; Phetteplace, Johnson, & Seidl, 2001; Verge, 2008; Veysset, Lherm, & Bébin, 2010; Williams, Audsley, & Sandars 2006). An LCA of GHG emissions from beef cattle includes many emissions not directly emitted from the feedlot and not accounted for in current EPA monitoring frameworks. A comprehensive inventory of production inputs within a defined boundary is required to estimate all GHG emissions, which includes impacts occurring away from facilities. For example, most LCAs of livestock would account for GHG emissions from the cropping system that provides cattle feed (Liska et al., 2009). Yet, a standard LCA boundary for GHG emissions from beef cattle is currently non-existent, which necessitates an extensive investigation of possible significant emissions that may occur either directly or indirectly from production.

Whereas LCAs of beef GHG emissions are not currently used in policy, there is also substantial interest in beef GHG emissions for labeling the environmental impact of products, as many production and retail companies wish to label "greener" products to inform consumers or to add retail value (Fliegelman, 2010).

1.5 Beef Production Systems

Beef cattle are dispersed throughout the U.S. to utilize available resources of forage and feed grains. Cattle move frequently throughout their life cycle due to the varied geography of cow-calf systems, feedlots, and processing locations. Calf populations (including dairy steers) are largest in Texas, Missouri, California, and Oklahoma, which accounted for ~25% of supply in 2003 (Shields & Mathews, 2003). Roughly half of the U.S. beef cow inventory is on rangeland and pastures on the Great Plains, and most cattle on feed are concentrated in the Central and Southern Plains where feed grains are abundant. Beef cattle feedlots in Texas, Kansas, Nebraska, and Colorado accounted for 65% of cattle on feed in 2003 (Shields & Mathews, 2003).

1.5.1 Beef Life Cycle Phases

The U.S. EPA defines three separate life cycle phases for cattle production: 1) calves, 2) replacements and stockers, and 3) feedlot animals. Most beef cattle (74%) are born between February and May, while dairy cattle are calved year round (EPA, 2010b). At seven months, a designated number of beef and dairy heifers are chosen as "replacements" for breeding and milking, while steers and remaining heifers are fed on pasture for ~0-17 months, depending on regional and temporal factors (EPA, 2010b; USDA, 2012a), and then transferred to feedlots. Time in feedlots depends on starting weights, rates of daily gain, and finishing weights, each of which depends on climate, feed sources, management, and other factors.

1.6 Review of Nitrogen Cycling Mechanisms

Livestock play a significant role in the global nitrogen cycle, directly and indirectly through use of forage and grain crops. This necessitates a basic understanding of global nitrogen cycling processes and nitrous oxide (N_2O) production. Most of the world's nitrogen is present in the atmosphere as inactive N_2; the atmosphere is 78.09% nitrogen (dry volume). However, there are two natural processes (biological nitrogen fixation and lightning), as well as two artificial processes (fossil fuel combustion and the Haber-Bosch process) which convert N_2 to reactive N. Haber-Bosch is an industrial process which typically uses natural gas to supply the energy and H_2 needed to produce ammonia (NH_3) from N_2 (Smil, 2001). All nitrogen (whether natural or artificial) must be returned to the environment at some point; this occurs through a variety of processes, in varying amounts and time periods (Figure 1). Of the 170 Tg N yr^{-1} applied to global cropland, 70% is lost to the environment, 20% is fed to animals, and 10% is fed to humans; of the 20% fed to animals, 17% is lost to the environment and only 3% is fed to humans as an animal product (Steinfeld, Mooney, & Schneider, 2010). Despite the relatively low efficiency of feed conversion by ruminants (e.g. cattle) relative to non-ruminants (e.g. swine and poultry), the nitrogen efficiency of beef feeding is generally higher than for non-ruminants since most pastures need no artificial fertilizer (Steinfeld et al., 2010). However, the precise nitrogen impact of beef is difficult to assess since the distinction between artificial and natural N is often arbitrary, metrics are difficult to define, and other byproducts besides meat, such as milk and eggs, have nitrogen cycle impacts.

Nitrogen flow to the environment is often coupled with transformations between oxidation-reduction states of nitrogen which are often mediated by microbes. Nitrogen in soil (often in the form of ammonium (NH_4^+)), proteins, and nucleic acids is generally reduced at the -3 level and is produced from N_2 by nitrogen fixing bacteria (or the Haber-Bosch process) or from NO_3^- via nitrate reduction via bacteria. Within the soil, organic materials are converted to ammonium through ammonification (also called nitrogen mineralization). Nitrification occurs when NH_4^+ is oxidized to NO_3^- (a +5 oxidation state) by chemoautotrophic bacteria in the *Nitrobacteraceae* family, as well as other heterotrophic organisms. Organic and ammonium forms of nitrogen are generally stable within soil; most N loss occurs through leeching of water-soluble NO_3^- (which has low anion-exchange capacity with soil) and volatilization of ammonium (NH_4^+) to the volatile gas ammonia (NH_3) (which occurs at high pH and in dryer soil) (Connor, Loomis, & Cassman, 2011). Volatized ammonia can then be redeposited on soil surfaces and subsequently nitrified/denitrified biologically to N_2O and other compounds (IPCC, n.d.). Additional loss occurs via denitrification, a process by which NO_3^- is released to the atmosphere, where each step is carried out by various heterotrophic bacteria including species from the genera *Pseudomonas, Bacillus, Thiobacillus, and Propionibacterium* in the following progression: $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$ (Connor et al., 2011). The principle end production is N_2, but often conditions allow for emission of the intermediate N_2O. Denitrification is favored by wet, anaerobic conditions and an abundant supply of nitrate and available carbon.

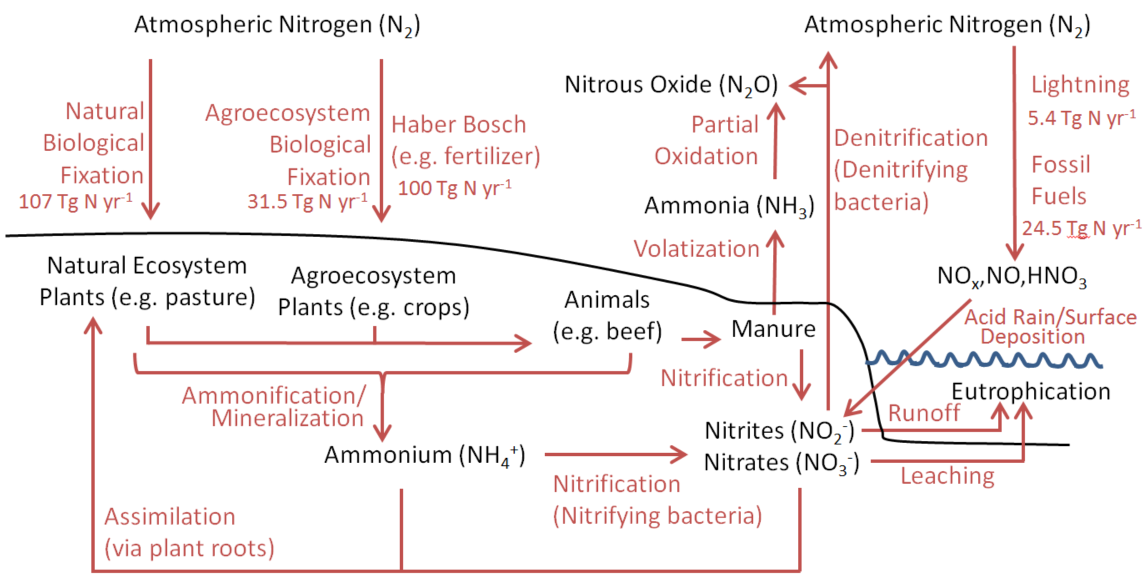
Estimates of livestock's impact are difficult to quantify since it is impossible to track reactive nitrogen as it is input into aquatic and terrestrial ecosystems before being denitrified into the atmosphere as N_2 and N_2O. By one estimate, of 17.7 Tg N year^{-1} of global N_2O emissions, agricultural soils are estimated to give off 6.3 Tg N year^{-1} of N_2O, where 2.1 is directly from animal waste management systems while synthetic fertilizer (0.9), animal waste (0.6), crop residue (0.4), leaching/runoff (1.6), atmospheric deposition (0.3) and other sources (0.4) account for the rest (Nieder & Benbi, 2008, p.

279). Other studies count cattle and feedlots as contributing 1.0 Tg N year^{-1} out of 14.9 Tg N year^{-1} of global N_2O (Nieder & Benbi, 2008, p. 279). Others estimate that industrialized animal production systems emit 1.8 Tg (40%) of the 4.5 Tg N-N_2O emitted globally to the atmosphere (Steinfeld et al., 2010, p. 93).

Many environmental factors affect the emission of N_2O. N_2O emission rate is generally linearly correlated with N application rate (Nieder & Benbi, 2008). Increasing soil water content increases denitrification; reducing O_2 concentration also increases N_2O production (this can be influence by soil texture, tillage, and water content). N_2O increase with increasing soil temperature and is positively correlated with higher levels of organic carbon, as well as in neutral or slightly alkaline soils. No-tillage increases denitrification due to higher C levels in topsoil and lower aeration. Conversion of land from forest and grasslands to cropland generally increase emission of nitrogen oxides while grazing increases N_2O from grasslands due to greater availability of inorganic N, as well as concentrated urine and dung patches.

Within the IPCC and EPA nitrous oxide characterization methods, direct N_2O emissions refer to nitrification and denitrification of manure and urine. Emissions are most likely to occur in dry manure handling systems that have aerobic conditions (favoring nitrification) with pockets of anaerobic conditions due to saturation (favoring denitrification). Additionally, indirect N_2O refers to two fractions of nitrogen losses: volatilization of ammonia (with redeposition and nitrification/denitrification to N_2O) and runoff/leaching (with denitrification to N_2O) (IPCC, n.d.; EPA, 2010a).

Figure 1. Components of the global nitrogen cycle



Fixation amounts are for early 1990s. Adapted from Steinfeld et al., 2010, p. 86.

1.7 Objectives

This introduction notes many of the background information necessary to understand the complexity of the beef production system. Throughout this study, our objectives here are to use quantitative methods to evaluate the beef life cycle in the following ways: use industry data to investigate quantitative differences in current EPA assessment methods, compare these with LCA methods, and determine the contribution of LCA components to total GHG emissions.

Chapter 2: Existing GHG Monitoring Frameworks: EPA Methods

2.1 Feedlot Data from Professional Cattle Consultants

To characterize U.S. dry feedlots (e.g. those without bedding or confinement), proprietary data from the Professional Cattle Consultants (PCC) published in monthly newsletters was compiled and analyzed (PCC, 2010). Data from the PCC are defined by five U.S. cattle regions, and in this study, data from the North Plains, Central Plains, and Corn Belt are used, comprising 11,575,000 steers and 9,635,000 heifers (Figure 2, Table 3). The North Plains region includes the state of Wyoming, and parts of Colorado, Kansas, Montana, Nebraska, North Dakota, and South Dakota. The Central Plains region includes parts of the states of Colorado, Kansas, New Mexico, Oklahoma, and Texas. The Corn Belt region includes the state of Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, Wisconsin, and parts of Kansas, Nebraska, North Dakota, and South Dakota. These surveys included in weight, out weight, average weight, days on feed, average daily gain, and dry matter intake (Table 3).

Figure 2. Map of the Professional Cattle Consultants (PCC) regions in the central U.S.

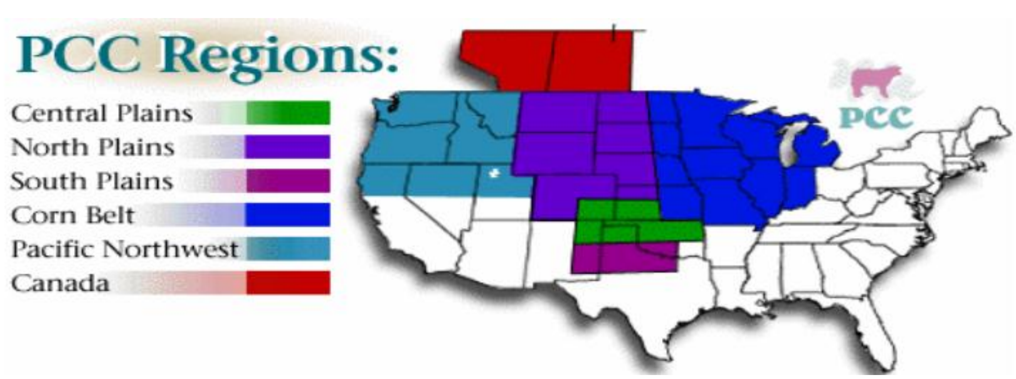


Table 3. Beef cattle regional performance data from the Professional Cattle Consultants.

		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	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
Plains	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	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
Belt	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	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
Plains	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	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
Average	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 & Galyean, 2007) less energy for maintenance (7.52 Mcal day^{-1}) (NRC, 2000).

Additional parameter values were used from the literature to describe feedlot performance, including energy content of feed dry matter (Vasconcelos & Galyean, 2007) and energy for maintenance (Table 6) (NRC, 2000). To characterize geospatial variability in feedlots, data from the EPA were used for animal mass, volatile solids, excreted nitrogen, ambient temperature, methane conversion factor, and fraction of nitrogen runoff/leaching (EPA 2010b). The total number of cattle by state was used to obtain a weighting factor for each parameter for each PCC region (USDA-NASS, 2012a). (Table 4). Due to limited data, variables such as crude protein in the diet, fraction of gross energy converted to methane, ratio of net energy for maintenance to digestible energy, etc. were held constant throughout the analysis (Table 6).

Industry data from these sources will be used within the Mandatory Reporting, Annual Inventory, and Life Cycle Assessment sections of this analysis; see tables associated with each section for specific values and assumptions.

Table 4. 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}) Volatile Solids		N excreted		Annual Inventory (kg animal^{-1} year^{-1}) Volatile Solids		N excreted		Ambient Avg temp[d] (°C)
				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 2 were analyzed with Image J software, ratio of pixels was compared
[b] (16)
[c] Texas, Percent of cattle in Central Plains is approximately 80% (based on approximation based on data from (16)); Avg temp is for Amarillo,TX to better represent northern region
[d] http://www.esrl.noaa.gov/psd/data/usclimate/tmp.state.19712000.climo
[e] (14)

2.2 EPA Mandatory Reporting

To gather GHG emissions data relevant for future climate policy, the Mandatory Reporting rule attempts to quantify all facility-level sources of emissions greater than 25,000 Mg CO_2 per year. The following equations define the approach (EPA, 2009).

Methane from Manure Management:

$$Equation\ JJ-2 = CH_4 Emissions_{Manure\ Management\ Systems}\ (\frac{metric\ tons}{year})$$
$$= \sum \left[(TVS_{AT} \times VS_{MMSC} \times (1 - VS_{SS}) \times \frac{365\ days}{1\ year} \times \frac{0.33\ m^3 CH_4}{kg\ VS\ added} \times MCF_{MMSC}) \times \frac{0.662\ kg\ CH_4}{m^3} \times \frac{1\ metric\ ton}{1000kg} \right]$$

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:

$$Equation\ JJ - 15 = Total\ Emissions\left(\frac{metric\ tons\ CO_2e}{year}\right) = [(CH_4 emissions_{MMS} + CH_4 emissions_{AD}) \times 21] + [Direct\ N_2O\ emissions \times 310]$$

For beef cattle feedlots in the North Plains, Central Plains, and Corn Belt, relatively consistent results occurred when data from the EPA, PCC, and American Society of Agricultural and Biological Engineers (ASABE) were applied to Mandatory Reporting equations and assumptions (Figure 3; Table 5). The use of PCC industry assumptions produced an emissions average for the combination of steers and heifers at 300 kg CO_2e per head per year, with large variability due to animal mass and excreted nitrogen ranging from 290 to 433 kg CO_2e per head per year. Comparatively, the average using EPA values was 526 kg CO_2e $hd^{-1}\ yr^{-1}$ and ASABE data values totaled 600 kg CO_2e $hd^{-1}\ yr^{-1}$ (Table 5). Using industry values, emissions from heifers (286 kg CO_2e $hd^{-1}\ yr^{-1}$) were lower than for steers (314 kg CO_2e $hd^{-1}\ yr^{-1}$) due to steers weighing more. Conversely, EPA values indicate heifers (542 kg CO_2e $hd^{-1}\ yr^{-1}$) emit more than steers (510 kg CO_2e $hd^{-1}\ yr^{-1}$) as heifers excrete more volatile solids (Table 5). Variability in emission levels between the three regions was minimal, being less than 6% for industry and 3% for EPA. Sensitivity analysis shows that the nitrogen excretion rate has the greatest influence on final emissions (Figure 3). Using industry data, gas contributions to total emissions from manure management as assessed by the Mandatory Reporting rule were found to be ~93% N_2O and ~7% CH_4.

Table 3. EPA Mandatory Reporting of GHG emissions, including manure only.

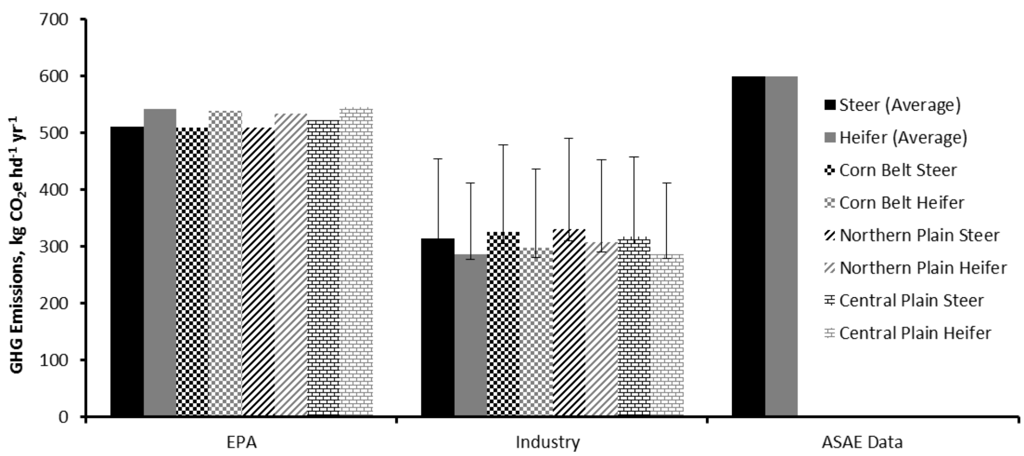
Parameter/Emission	Units	EPA Assumptions Steer	EPA Assumptions Heifer	Industry Assumptions Steers	Industry Assumptions Heifers	ASAE Data Avg.
Methane						
TAM_{AT}	kg head^{-1}	420[a]	420	459.3[b]	418.6	446[c]
MCF_{MMSC} [d]	decimal	1.14%	1.14%	1.14%	1.14%	1.14%
VS_{AT}	kg VS day^{-1} 1000kg^{-1}	3.97[e]	4.35	1.58[f]	1.58	4.25[c]
Total CH_4 emissions[g]	kg CO_2e hd^{-1} yr^{-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} yr^{-1}	473.8	502.5	298.3	271.9	558.6
Total GHG emissions	kg CO_2e hd^{-1} yr^{-1}	510.1	542.4	314.1	286.3	599.9
Total GHG emissions (average)	kg CO_2e hd^{-1} yr^{-1}	526.3		300.2		

[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^{o}) and 0.5% (temperate ambient temp = 15-25 C^{o}), 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, (BFNMP$, 2009)
[g] Equations JJ-2 and JJ-3 (EPA, 2009).
[h] Equations JJ-13 and JJ-14 (EPA, 2009).

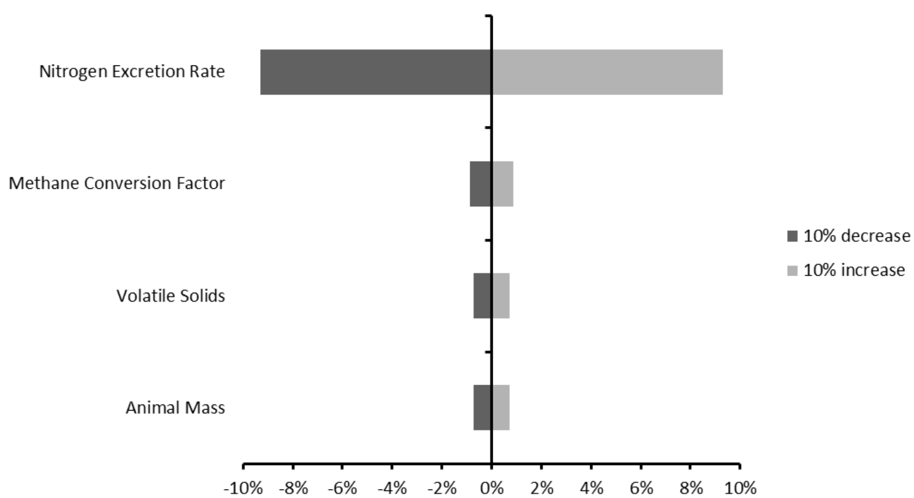
Additional assumptions:
VS_{SS} (VS removal by solids separation) = 0 (no solid separation)
B_o (Maximum CH_4 conversion factor) = 0.33. Table JJ-2 (assume feedlot steers) (EPA, 2009).
VS_{MMSC} (fraction manure in system) = 1 (assume all manure is in dry lot feedlot)
EF_{MMSC} = 0.02 kgN_2O-N/kgN_2O.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

Figure 3. EPA Mandatory Reporting Methodology for quantification of cattle greenhouse gas emissions (kg CO_2e hd^{-1} yr^{-1}): a) Comparison of results b) Sensitivity of parameters using EPA data. See Table 5.

a)



b)



2.2.1 Consistency with EPA thresholds

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 (EPA, 2009, p. 56485). Yet emissions for a representative 30,000 head feedlot that uses the above assumptions totals 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}) for the 30,000 head feedlot. 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.

2.3 EPA Annual Inventory

To estimate national GHG emissions, the EPA Annual Inventory consists of four components: CH_4 from enteric fermentation, CH_4 from manure, direct N_2O from manure volatilization, and indirect N_2O from runoff and leaching with subsequent volatilization (Table 6; Figure 4). The following equations are key to deriving this approach (EPA, 2010b).

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):

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

The PCC values formed the basis for an industry emissions estimate of 1653 kg CO_2e hd^{-1} yr^{-1}, ranging from 1590 to 1716 kg CO_2e hd^{-1} yr^{-1}. Two general annual emissions calculations are possible, one where the EPA uses default values for volatiles solids (VS) and $N_{excreted}$ parameters and totals 1611 kg CO_2e hd^{-1} yr^{-1}. Alternatively, similar calculations by the IPCC (which was the original basis for the EPA methods (EPA, 2010b, p. A-122), use empirical equations to determine both VS and $N_{excreted}$, where these calculations sum to 1613 kg CO_2e hd^{-1} yr^{-1}. ASABE values used in EPA calculations correspond to 1668 kg CO_2e hd^{-1} yr^{-1}. Using EPA calculations, steers were calculated as emitting ~60 kg CO_2e hd^{-1} yr^{-1} more than heifers on an annual basis. Spatial differences between the three PCC regions were 0.3% for values from using EPA methods and values, 0.4% for EPA methods and IPCC values, and 5% for EPA methods and industry values. In a general comparison of the sensitivity of five parameters (a change of ±10% for 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 4). 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 (Table 6).

Table 4. 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]	$kgCO_2e\ hd^{-1}\ yr^{-1}$	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]	$kgCO_2e\ hd^{-1}\ yr^{-1}$	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]	$kgCO_2e\ hd^{-1}\ yr^{-1}$	519.6	508.0	563.7	529.6	652.3	652.3	652.3
Indirect N_2O Emissions[u]	$kgCO_2e\ hd^{-1}\ yr^{-1}$	64.3	62.9	69.8	65.6	80.8	80.8	80.8
Total GHG emissions	$kgCO_2e\ hd^{-1}\ yr^{-1}$	1640.0	1583.0	1655.3	1571.7	1688.5	1618.1	1679.9
Total GHG emissions (average)	$kgCO_2e\ hd^{-1}\ yr^{-1}$	1611.5		1613.5		1653.3		

[a] Table A-171, assume feedlots and year 2009 (EPA, 2010b)
[b] (PCC, 2010)
[c] (ASABE, 2010)
[d] Page A-206, 2.8 to 3.3 lbs day^{-1} (EPA, 2010b)
[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 & Galyean, 2007). Subtract NE_m to get NE_g (Table 3)
[i] Page A-212 in Section 3.9 (EPA, 2010b)
[j] DayEmit equation. Page A-212 (EPA, 2010b)
[k] Table A-186 (assume On Feed Beef Steer, Nebraska), cited from Moffroid and Pape, 2010 (EPA, 2010b)
[m] see equation, Page A-216 (refers to IPCC2006 Tier II equations), assume UE=.02*GE for feedlot, assume ash content = .08 (EPA, 2010b)
[n] assume 85% digestibility (BFNMP$, 2009).
[p] Equation, Page A-222 (EPA, 2010b)

[q] Total Kjeldahl N excretion rate, Table A-186 (assume On Feed Beef Steer, averaged over regions), cited from Moffroid and Pape, 2010 (EPA, 2010b)
[r] Equations, Page A-217 (EPA, 2010b) based on IPCC 2006, Tier II equations and constants, assume percent crude protein = 13.34% (Vasconcelos & 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) (BFNMP$, 2009).
[t] Equation, Page A-223 (EPA, 2010b)
[u] Equation, Page A-224 (EPA, 2010b)

Assumptions

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

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, 2010b)

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

Table A-187, assume dry lot

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

Methane Conversion Factor (MCF)=0.11, Table A-189, assume aerobic treatment and weighted average over central U.S. (Table 4) (EPA, 2010b)

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, 2010b)

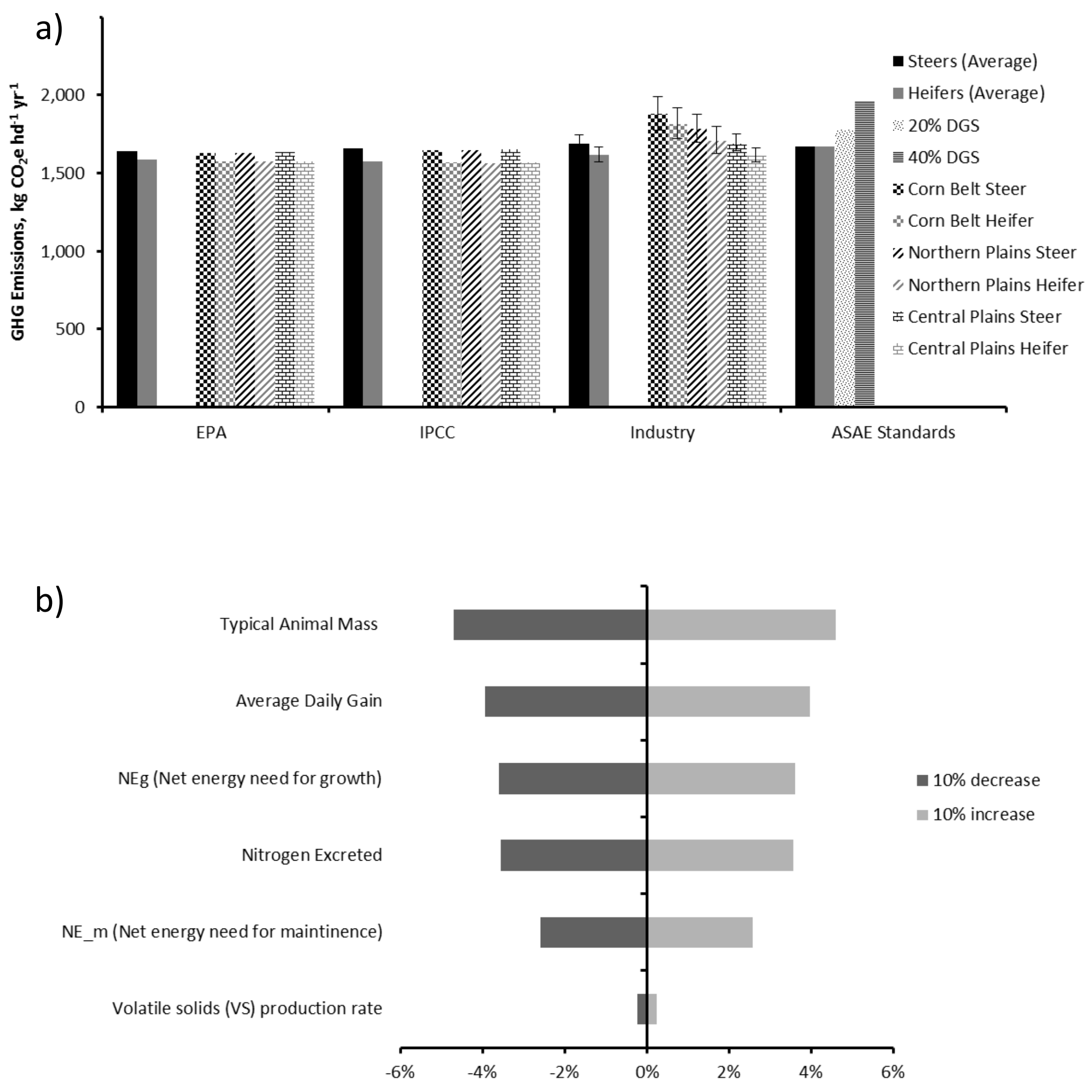
$EF_{volitalization}$ = .010 kg N_2O-N/kg N. Indirect N_2O emission factor for volatilization, page A-224 (EPA, 2010b)

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

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

$Frac_{runoff/leach}$ = 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, 2010b)

Figure 4. EPA Annual Inventory Methodology for quantification of cattle greenhouse gas emissions (kg CO_2e hd^{-1} yr^{-1}): a) Comparison of results b) Sensitivity of parameters using EPA data. See Table 6.

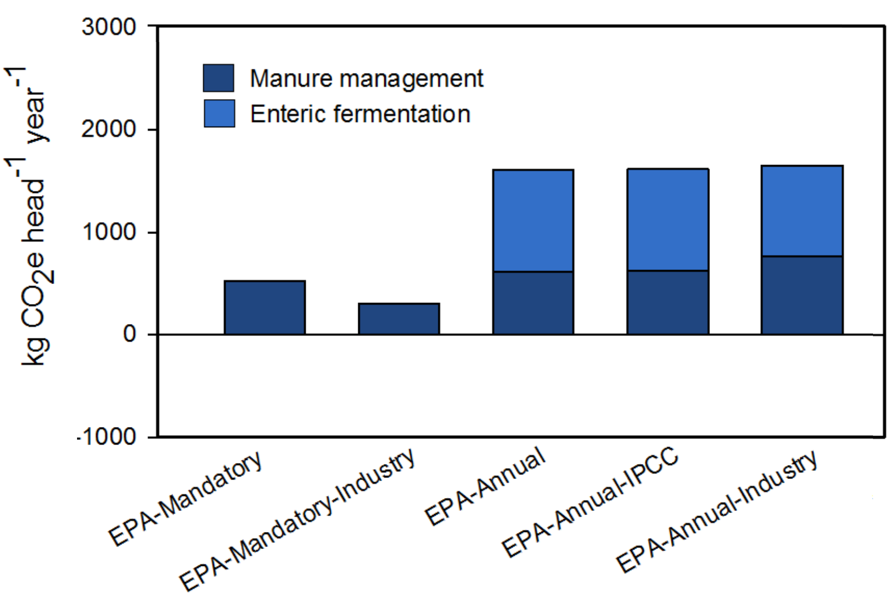


2.4 EPA System Boundaries

The EPA Mandatory Reporting and Annual Inventories only estimate localized GHG emissions from the feedlot portion of cattle production. Feedlots would be the only part of the production sequence where GHG emissions could be concentrated enough to meet the regulatory threshold in the EPA's Mandatory Reporting methodology (25,000 Mg CO_2e per year, designated by the EPA as 29,300 head, encompassing an estimated 50 operations in the U.S.) (EPA, 2009; EPA 2012b). The Annual Inventory methodology was developed to comply with international agreements and is based on IPCC methods; parameters such as volatile solids production rate and nitrogen excretion rate were calculated using the original IPCC equations (IPCC, 2006)(Table 6). The EPA equations used in Mandatory Reporting and Annual Inventories were previously published (EPA, 2009; EPA, 2010b). As intermediate metrics, both the Mandatory Reporting and Annual Inventory use kg CO_2e per head per year, but both ultimately produce results in Mg CO_2e per year for a feedlot facility and national industry, respectively.

The Mandatory Reporting only includes emissions from manure management, and the Annual Inventory includes both manure management and enteric fermentation. A graphical comparison is shown of these boundary differences, which are limited to direct emissions from the feedlot portion of the cattle life cycle (Figure 5).

Figure 5. Beef cattle feedlot GHG emissions from EPA methodologies and system boundaries. *EPA Mandatory* uses Mandatory Reporting boundaries and includes only data from EPA documentation (EPA, 2009; EPA, 2012b) while *EPA Mandatory-Industry* uses PCC data (2010; Table 5). *EPA-Annual* utilizes EPA Annual Inventory boundaries and values (EPA, 2010b), *EPA-Annual-IPCC* uses IPCC empirical equations in conjunction with EPA values (IPCC, 2006; Vasconcelos & Galyean, 2007), and *EPA Annual-Industry* employs several industry sources (NRC 2000; PCC, 2010) (Table 6).



Chapter 3: Life Cycle Assessment of Beef Cattle GHG Emissions

3.1 Life Cycle Assessment Principles and Limitations

Life Cycle Assessment (LCA) is defined as "a technique for assessing the environmental aspects and potential impacts associated with a product" (ISO 14040, i). It is an iterative process in which practitioners define a goal and scope, conduct an inventory analysis, assess impacts, and interpret results. LCA is a valuable tool, however, it also has inherent limitations.

The International Standards Organization (ISO) provides an outline of LCA methodologies, though it readily acknowledges that it provides only an outline and not a detailed procedure. It also notes several important considerations:

- "If LCA is to be successful in supporting environmental understanding of products, it is essential that LCA maintains its technical credibility while providing *flexibility, practicality,* and *cost effectiveness* of application" (ISO, 2006, p. iii).
- "the nature of choices and assumptions made in LCA (e.g. system boundary setting, selection of data sources, and impact categories) may be *subjective*" (ISO, 2006 p. iv).
- "Results of LCA studies focused on global and regional issues may not be appropriate for local applications" (ISO, 2006, p. iv).
- "The lack of spatial and temporal dimensions in the inventory data used for impact assessment introduces *uncertainty* in impact results." (ISO, 2006, p. iv).

In summary, comparing the results of different LCA studies is only possible if the assumptions and context of each study are identical. These assumptions should be explicitly stated for reasons of transparency (ISO, 2006, p. iv). It should also be noted that “there is no scientific basis for reducing LCA results to a single overall score or number since trade-offs and complexities exist for the systems analyzed at different stages of their life cycle.” (ISO, 2006, p. 4).

3.1.1 Attributional versus Consequential LCA

Theoretically, two broad categories of LCA exist: attributional and consequential. “Attributional LCA (ALCA) provides information about the impacts of the processes used to produce (and consume and dispose of) a product, but does not consider indirect effects arising from changes in the output of a product” (Brander et al., 2008). In other words, ALCA is a static “total emissions” approach and is characteristic of the methods discussed so far as well as consistent with principles from ISO standards. On the other hand, “Consequential LCA (CLCA) provides information about the consequences of changes in the level of output (and consumption and disposal) of a product, including effects both inside and outside the life cycle of the product” (Brander et al., 2008). CLCAs are useful for policy discussions and examining the causal relationships between dynamic economic and logistical changes. “Whereas ALCAs are generally based on stoichiometric relationships between inputs and outputs, and the results may be produced with known levels of accuracy and precision, CLCAs are highly dependent upon economic models representing relationships between demand for inputs, price elasticities, supply, and markets effects of co-products. Such models rarely provide known levels of accuracy or precision and should therefore be interpreted with caution" (Brander et al., 2008).

It is a common misconception that "direct" emissions correlate to ALCA and "indirect" emissions are for CLCA and that the two can be summed. This is not possible as there are key differences between the methods; it is very likely that double counting would occur since coproducts are treated differently. It is also helpful to note that CLCAs could potentially be negative (for example, increasing dairy production could displace meat from beef feedlots). In sum, CLCA should ideally be used for policy analysis and decision making when insights outweigh the uncertainty; ALCA is designed for product comparison and GHG inventory construction. Brander et al. 2008 even suggest that LCA should not be used at all and an alternate, pre-decided indicators for the success or failure of a policy that is readily monitored.

Many LCAs exist in literature, yet few differentiate themselves between the attributional and consequential methodologies. Indeed many of the regulatory LCAs are inconsistent hybrids (i.e. the California Air Resources Board LCAs include both emissions from fuel burning (attributional) and land use change (consequential)) (Sanchez et al., 2012). Understanding the difference between these two types is important, especially since regulators are inconsistently combining the approaches.

3.2 GHG Emissions from Feed Production and Direct Feedlot Sources

An attributional beef feedlot LCA would also include emissions from all direct and off-site inputs and outputs. The boundaries for the feedlot LCA presented here are limited to inputs and outputs during the feedlot operation, and do not consider the pasture cow-calf stage, which is later included in the complete LCA of beef production. The LCA includes enteric fermentation and manure management emissions as described by Annual Inventory and IPCC methodology (EPA, 2010a). Additionally, emissions from corn grain and alfalfa production, manure as an off-site soil carbon amendment, and feedlot fossil fuel use were used. An attributional approach for quantification of land use change (LUC) GHG emissions from grain consumption is included as well. Emissions per amount of usable product are the standard units for LCA; thus, in the full LCA, $kgCO_2e\ kg^{-1}$ beef replaces $kgCO_2e\ hd^{-1}\ yr^{-1}$ as the units of comparison.

Direct GHG emissions from beef feedlots include fossil fuel use in management operations (144 $kgCO_2e\ hd^{-1}\ yr^{-1}$) (Steinfeld et al., 2006), and changes in soil organic carbon from spreading manure (-124 $kgCO_2e\ hd^{-1}\ yr^{-1}$) (Fronning et al., 2008). A carbon sequestration rate of 106 kg C $ha^{-1}\ yr^{-1}$ was assumed (Fronning et al., 2008), however higher values (200-500 kg C $ha^{-1}\ yr^{-1}$) have been reported (Follett, 2001). In addition to the direct GHG emissions from the beef feedlot, attributional LCA quantifies related production emissions that occur away from the EPA-defined facility. Beef cattle are fed primarily corn grain which is produced in energy and GHG intense field operations and contribute 788 $kgCO_2e\ hd^{-1}\ yr^{-1}$ to the feedlot LCA (Liska et al., 2009) (Figure 6; Table 7).

3.3 Land Use Change

A consequential LCA accounts for emissions that occur from a range of sources that may change as a consequence of production (Finnveden et al., 2009). Indirect land use change (ILUC) was recently estimated to occur based on the change in level of demand for corn for ethanol production, and a similar calculation is employed in state (California) and federal (EPA's Renewable Fuel Standard, RFS2) regulatory LCAs for corn-ethanol (Hertel et al., 2010; Liska & Perrin, 2009; Searchinger et al., 2008; Wang et al., 2011). Economic analysis estimated the marginal change in grain price due to the change in grain demand, and the accompanied 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 growth of cropping areas.

If the U.S. beef cattle population were increasing, such an additional GHG emission from ILUC 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 the ILUC calculation. The cattle population cycles due to various factors, but for this analysis it is assumed 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-NASS, 2012a).

A second approach to calculating GHG emissions from land use change (LUC) 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 LUC directly from the rate of grain consumption (Flysjö, Cederberg, Henriksson, & Ledgard, 2011). Feed for most European cattle is sourced from LUC sensitive areas (e.g. Brazil), which prompts European LCA practitioners to focus on physical relationships between cattle and LUC. On the other hand, nearly all feed for U.S. cattle is sourced from within the United States; this means US LUC relationships are primarily due to economic teleconnections and require a broader set of life cycle impact boundaries. In light of this, a more 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, Brander, Chatterton, Murphy-Bokern, Webster, & Williams, 2009). By this method, it is assumed that 1.43 Mg CO_2 is emitted from LUC from each hectare of agricultural land used (including crop and pasture) regardless of usage; this value attributes 58% of global land use change emissions to agriculture. To determine the LUC impact of the feedlot, the amount of land necessary for sufficient corn production (per head) was multiplied by the LUC emissions factor. Within the pasture phase of the life cycle, the LUC impact was determined by multiplying the LUC emissions factor by the amount of pasture land utilized by each cow.

By including emissions from crop, urea, and alfalfa production (863 $kgCO_2e\ hd^{-1}\ yr^{-1}$) and land use change based on the area of land used per head (582 $kgCO_2e\ hd^{-1}\ yr^{-1}$), GHG emissions from feedlot beef cattle nearly double compared to the direct feedlot emissions based on the Annual Inventory assessment (1674 $kgCO_2e\ hd^{-1}\ yr^{-1}$)(Figure 6, Table 7).

There is currently no consensus on how to employ LUC to livestock. However, there have been explicit calls for LUC to be included in beef LCAs (Garnett, 2009). By including LUC based on a global average rate of agricultural land use, a rough LUC has been estimated here, but it is likely associated with a large uncertainty. This value only is applied to estimate the relative magnitude of LUC compared to other factors, and to estimate a maximum level of life cycle GHG emissions from beef cattle. If the EPA were to monitor cattle production using LCA methods, it is reasonable to assume that LUC emissions would not be counted as they have been for corn-ethanol, because the cattle population is not changing (Hertel et al., 2010; Liska & Perrin, 2009; Searchinger et al., 2008; Wang et al., 2011). It is clear that a general LCA perspective could apply LUC based on continuous demand as shown here, but this is perhaps unlikely to be used by the EPA. When LUC and cropping emissions were added to the Annual Inventory, total emissions per head per year more than doubled.

3.4 Ethanol Coproducts Use in Feedlots

Corn ethanol production has risen dramatically in recent years in the central U.S., resulting in expanded use of coproducts as livestock feeds (Bremer et al., 2010). Distillers grains plus solubles (DGS) have a higher energy density than the corn it displaces, resulting in greater daily gain and less time in the feedlots or greater beef production compared to conventional corn diets (Table 7). As DGS contain a larger fraction of protein, nitrogen from urea is not added to DGS-supplemented diets. On average, DGS are fed at 20% of dry matter intake when substituted in corn-based beef cattle diets (Bremer et al., 2010)**.** With growing ethanol production, DGS could be substituted into cattle diets at a 45% of dry matter intake. When considering DGS use in the LCA of feedlot cattle, the decreased feeding time was calculated her to reduce GHG emissions per head from 1640 $kgCO_2e\ hd^{-1}$ (conventional diet) to 1548 $kgCO_2e\ hd^{-1}$ (current inclusion) to 1730 $kgCO_2e\ hd^{-1}$ (maximum inclusion) (Table 7). The GHG emissions from DGS are assumed to be identical to corn grain by mass (changes in emissions from drying coproducts were not assessed). Thus, feeding DGS reduces GHG emissions from a beef cattle feedlot by 5.7% at current levels, but increases emissions by 5.4% at future DGS levels due to higher nitrogen excretion and estimated N_2O emissions. If only wet DGS were fed at the maximum inclusion level to a subset of local feedlots, the resulting emissions would be roughly equivalent to a conventional corn diet (Table 7). Yet, if pasture emissions (7.97 $kgCO_2e\ kg^{-1}$ beef) (Pelletier et al., 2010) from the cow-calf system are included in the beef life cycle, the current DGS diet (16.26 $kgCO_2e\ kg^{-1}$) would reduce GHG emissions by 1.8% compared to the conventional corn-based diet (16.55 $kgCO_2e\ kg^{-1}$); whereas higher DGS diets (16.90 $kgCO_2e\ kg^{-1}$) are estimated to increase emissions by 2.1% (Table 8).

Although feeding coproducts markedly changes the chemical composition of corn fed diets, data on changes in enteric fermentation are not available and thus no difference is assumed. Nitrogen in the diet is higher with coproducts, which results in an 18% increase in excreted nitrogen and N_2O emissions for current DGS inclusion levels over the conventional diet (Luebbe et al., 2012).

Figure 6. Beef cattle GHG emissions from different methodologies and system boundaries from feedlots. Feedlot LCA-Conventional uses the most inclusive system boundaries for the feedlot (see Table 7). See Figure 5 for other caption details. Error bars include 90% of Monte Carlo simulation values (Table 9; Table 10; Figure 7).

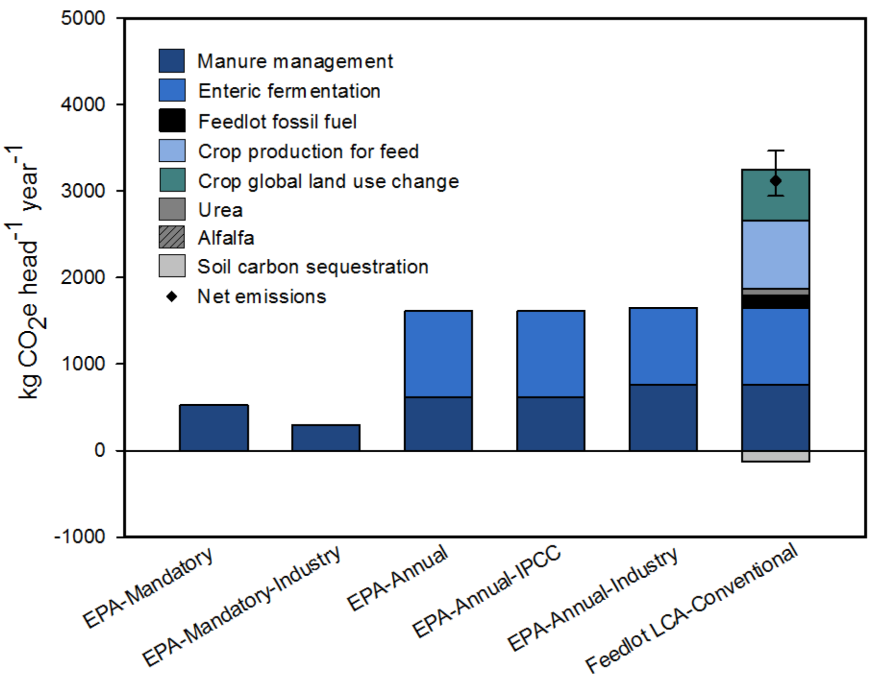


Table 5. Inventory of GHG emissions from U.S. beef cattle feedlots.

EPA Annual Inventory calculations (EPA, 2010b) with industry source data (PCC, 2010). The feed production scenarios Conventional Corn Diet, Current DGS Use, and Maximum DGS Use estimated default industry practice, current use of distillers grains plus solubles (DGS) from ethanol production, and hypothetical maximum DGS feeding rate, respectively (Bremer et al., 2010). The scenario Max Wet DGS Use uses wet DGS only.

Emissions Sources	**Units**	**Conv Corn Diet**	**Cur-rent DGS Use**	**Max DGS Use**	**Max Wet DGS Use**
Feedlot					
Manure Management[a]					
N-excretion rate	kg N hd^{-1} yr^{-1}	*69.5*	*82.2*	*103.8*	*103.8*
N_2O (direct & indirect)	$kgCO_2e$ hd^{-1} yr^{-1}	733	865	1092	1092
CH_4	$kgCO_2e$ hd^{-1} yr^{-1}	33	33	33	33
Enteric Fermentation (CH_4)[a]	$kgCO_2e$ hd^{-1} yr^{-1}	888	888	888	888
Feedlot Fossil Fuel Use[b]	$kgCO_2e$ hd^{-1} yr^{-1}	144	156	150	156
Cattle on farm in one year[c]	*head* yr^{-1}	*1.901*	*2.062*	*1.973*	*2.062*
Soil Organic Carbon from Manure[d]	$kgCO_2e$ hd^{-1} yr^{-1}	-124	-124	-124	-124
	$kgCO_2e$ hd^{-1} yr^{-1}	**1674**	**1818**	**2038**	**2045**
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]	*kg* hd^{-1} day^{-1}	*9.14*	*7.05*	*4.44*	*4.44*
Days on Feed[h]	*days*	*192*	*177*	*185*	*177*
Urea[h]	$kgCO_2e$ hd^{-1} yr^{-1}	71	-	-	-
Alfalfa hay[i]	$kgCO_2e$ hd^{-1} yr^{-1}	4.3	4.3	4.3	4.3
Corn Production[j]	$kgCO_2e$ hd^{-1} yr^{-1}	788	788	788	788
	$kgCO_2e$ hd^{-1} yr^{-1}	**863**	**792**	**792**	**792**
Land Use Change from Cropping[k]	$kgCO_2e$ hd^{-1} yr^{-1}	**582**	**582**	**582**	**582**
Total Feedlot Emissions	$kgCO_2e$ hd^{-1} yr^{-1}	**3119**	**3192**	**3412**	**3419**
Feedlot Emissions per head	$kgCO_2e$ hd^{-1}	**1640**	**1548**	**1730**	**1658**
Percent relative to conventional	%	*100*	*94.3*	*105.4*	*101.1*

[a] see Table 4, assume $N_{excreted}$ increased by 18% (for 15% WDGS scenario) and 49% (for 30% WDGS scenario) which correlates to current DGS and the maximum scenarios, respectively (Luebbe, Erickson, Klopfenstein, & Greenquist, 2012); similar conclusions from (Regassa, Koelsch, & Erickson, 2008)

[b] GHG Intensities: Diesel=0.047 ton CO_2e hd^{-1}, LPG= 0.015 ton CO_2e hd^{-1}, Electricity=0.014 ton CO_2e *hd^{-1}, Total= 0.076 ton CO_2e hd^{-1}*, Minnesota is characteristic of central U.S., see Table 3.10, page 101 (Steinfeld et al., 2006).

[c] inverse of days on feed (Bremer et al., 2010)
[d] Applying manure at 45 kg N ha^{-1} yr^{-1} and carbon sequestration at a rate of 106 kg C ha^{-1} yr^{-1} (Fronning, Thelen, & Min, 2008)
[e] Current DGS feed on average is composed of 24% dry DGS, 38% modified DGS, and 38% wet DGS; Maximum DGS scenario is composed of 62% dry DGS, 19% modified DGS, and 19% wet DGS (Bremer et al., 2010).
[f] Average dry matter feed intake of corn-based diet, 10.45 kg hd^{-1} day^{-1} (PCC, 2010) times inclusion level (Bremer et al., 2010).
[g] Average daily dry matter intake, times 87.5%, 67.5%, and 42.5%, respectively (Bremer et al., 2010).
[h] Urea intake=0.13 kg hd^{-1} day^{-1} &GHG Intensity=1.5 $kgCO_2e$ kg^{-1} urea yields 0.195 $kgCO_2e$ hd^{-1} day^{-1} (Bremer et al., 2010)
[i] Alfalfa intake=7.5% dm=0.78 kg hd^{-1} day^{-1}, GHG Intensity=31.1kgC ha^{-1}, Yield=3.4 short ton $acre^{-1}$ equates to 0.015 $kgCO_2e$ kg^{-1} dry matter which yields 0.012 $kgCO_2e$ hd^{-1} day^{-1}, assume 114 $kgCO_2$ ha^{-1} GHG intensity, based on fossil fuel use for one seedling year, two established years, and one final year (Adler, Del Grosso, & Parton, 2007), U.S. average yield of 3.4 tons $acre^{-1}$ in 2011 (USDA-NASS, 2012b) and feeding rate of 0.784 kg hd^{-1} day^{-1}, or 286 kg hd^{-1} yr^{-1} (Bremer et al., 2010).
[j] Corn and DGS are assumed to have same direct GHG intensity of 0.236 $kgCO_2$ kg^{-1} grain; calculated by spatial averaging over central U.S. (Liska et al., 2009), see Table 10 for values and weighting/frequency factors.
[k] LUC Emissions from Cropping = Land Intesity of corn * LUC GHG Intensity of agricultural land
Land intensity of corn = corn consumed / corn yield = 0.407 ha hd^{-1} where

Corn Consumed = 3337 kg hd^{-1} yr^{-1} assume constant 9.14 kg hd^{-1} day^{-1} daily intake of corn or DGS (Bremer et al., 2010)

Corn Yield = 8.20 Mg ha^{-1} yr^{-1}, (Bremer et al., 2010) Central U.S. average, see Table 10 for values and weighting/frequency factors.

LUC GHG Intensity, agricultural land = 1430 $kgCO_2$ ha^{-1} yr^{-1}, assume land use change intensity for all agricultural land to be 1.43 $MgCO_2e$ ha^{-1} (Audsley et al. 2009)

3.5 Complete Life Cycle Assessment of Beef Cattle

When meat production emissions are considered in LCA, the appropriate metric is emissions per mass of product. In addition to feed and feedlot emissions as defined above, feedlot operations also require a system to produce incoming feeder cattle for finishing to market weight. Prior to arriving at the feedlot, beef cattle are fed on pasture for roughly 0-17 months, depending on regional and temporal factors (Shields & Mathews, 2003). In pasture systems, GHG emissions occur primarily from enteric fermentation (~42%), feed production (~37%), and manure (~21%) from both cow and calf (Pelletier et al., 2010). When pasture emissions (7.97 kgCO_2e kg^{-1} beef), land use change from pasture (2.48 kgCO_2e kg^{-1} beef), and processing (0.15 kgCO_2e kg^{-1} beef) are added to the LCA emissions of the feedlot, total LCA emissions grow by nearly three-fold from 5.95 to 16.6 kgCO_2e kg^{-1} beef (Table 8).

Table 6. Life cycle assessment of GHG emissions from U.S. beef, including pasture and feedlot.

Emissions Sources	**Units**	**Conv Corn Diet**	**Current DGS Use**	**Max DGS Use**	**Max Wet DGS Use**
Feedlot Emissions[a]	$kgCO_2e\ kg^{-1}$ beef	***5.95***	***5.65***	***6.25***	***6.01***
Emissions from Pasture[b]	$kgCO_2e\ kg^{-1}$ beef	**7.98**	**7.98**	**7.98**	**7.98**
Emissions from Processing[c]	$kgCO_2e\ kg^{-1}$ beef	**0.15**	**0.15**	**0.15**	**0.15**
Land Use Change from Pasture[d]	$kgCO_2e\ kg^{-1}$ beef	**2.48**	**2.48**	**2.48**	**2.48**
LCA Total Net Emissions	$kgCO_2e\ kg^{-1}$ beef	**16.55**	**16.26**	**16.90**	**16.62**
Percent relative to conventional	%	*100*	*98.2*	*102.1*	*100.4*

[a] Conversion ($kgCO_2e\ hd^{-1}\ yr^{-1}$ to $kgCO_2e\ kg^{-1}$ beef) of feedlot emissions from Table 7), assumes slaughter weight of 584 kg, dressing percentage of 63% and 75% meat:waste ratio.

[b] 63% of life cycle emissions are from pasture (Pelletier et al., 2010).

[c] (Steinfeld et al., 2006)

[d] Assume:

Total Cattle = 32,800,000 head (USDA-NASS, 2012a) = sum of state counts for cattle, cows, beef inventory, 2007 Census of Agriculture

Pasture acres attributed to beef = 38,800,000 acres, derived from (USDA-NASS, 2012a) = 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" =

$$Total\ Pasture\ Acres\ \times \frac{109{,}900{,}000\ lb\ beef\ slaught.\times 1.91\frac{\$}{lb}}{109{,}900{,}000\ lb\ beef\ slaught.\times 1.91\frac{\$}{lb}+4{,}600{,}000\ lb\ lamb\ slaught.\times 1.34\frac{\$}{lb}}$$

Beef pasture density = 0.478 ha hd^{-1} = pasture acres attributed to beef/ total cattle

Pasture LUC emission per head = 683.9 $kgCO_2e\ hd^{-1}\ yr^{-1}$ = LUC GHG Intensity of agricultural land * beef pasture density

Land use change intensity for all agricultural land = 1.43 Mg ha^{-1} (Audsley et al., 2009) See Table 7 for similar land intensity calculations. For comparison, 7.98 $kgCO_2e\ kg^{-1}$ beef = 684 $kgCO_2e\ hd^{-1}\ yr^{-1}$

3.6 Monte Carlo Simulation and Uncertainty Analysis

Depending on a number of spatial and temporal factors that affect crop production and cattle performance, a range of GHG intensities for the LCA is expected to occur. To generate a probability distribution of the expected LCA intensities, Monte Carlo simulation, a stochastic method of repeated random sampling, was used. The program @Risk (Palisade Corporation, Ithaca, NY, www.palisade.com) was used to compute 10,000 iterations of outputs by varying six parameters in a manner consistent with their probability of occurrence. Three parameters (animal mass, daily gain, and energy for gain) were assigned a normal distribution consistent with a known standard deviation from PCC data and 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 9).

Table 7. 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	248.59	8.53	-

[a] See Table 3
[b] Dry matter intake determines energy for gain parameter. See Table 3
[c] MCF
[d] Methane Conversion Factor (MCF), Table A-189, assume aerobic treatment and weighted average over central U.S. (Table 4) (EPA, 2010b).Frequency determined by state averages, see Table 4
[e] (14)
[f] Determined by comparing levels of corn production for various states, average of years 2003-2005 (USDA-NASS, 2012a).

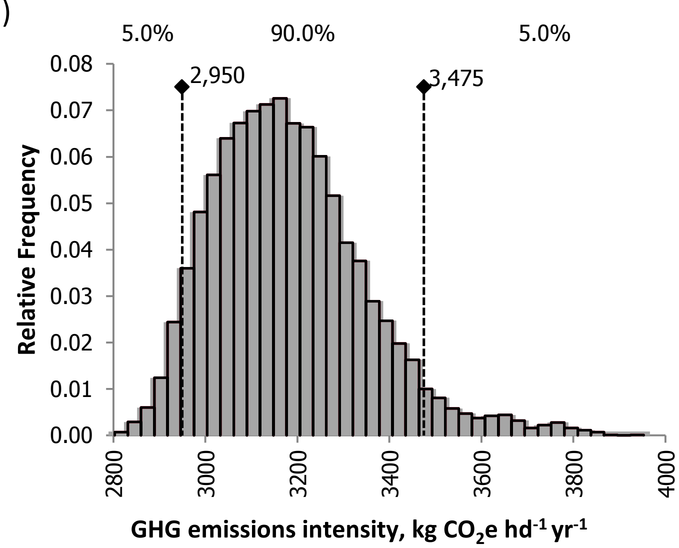
Available statistics were applied in a stochastic Monte Carlo simulation to generate probability distributions of GHG intensities for both the feedlot and the full LCA. Without inclusion of land use change, the mean and standard deviation of feedlot emissions intensity is 2673 ± 156 kg CO_2e hd^{-1} yr^{-1} (Table 10, Figure 7). Where land use change is included, emissions increase to 3182 ± 167 kg CO_2e hd^{-1} yr^{-1}. The corresponding GHG intensity of beef with the most inclusive life cycle boundaries, including pasture, is 16.67 ± 0.32 kg CO_2e kg^{-1} beef (Figure 7). This distribution is skewed to the right, due to the distribution of emissions from crop production, where the majority of production occurs at lower intensities, but many relatively inefficient states still produce corn at lower production levels (e.g. South Dakota) (Table 9). If land use change were not included in the life cycle inventory, the emissions intensity is 12.35 ± 0.32 kg CO_2e kg^{-1} beef. Quantifying these distributions of possible results is limited by lack of information concerning the distribution of most parameters; only six were tested in this analysis. The two factors that would likely further expand the range of possible results include emissions from pasture systems and those from land use change, but additional data for these factors is limited.

Table 8. Monte Carlo simulation probability results using @Risk.

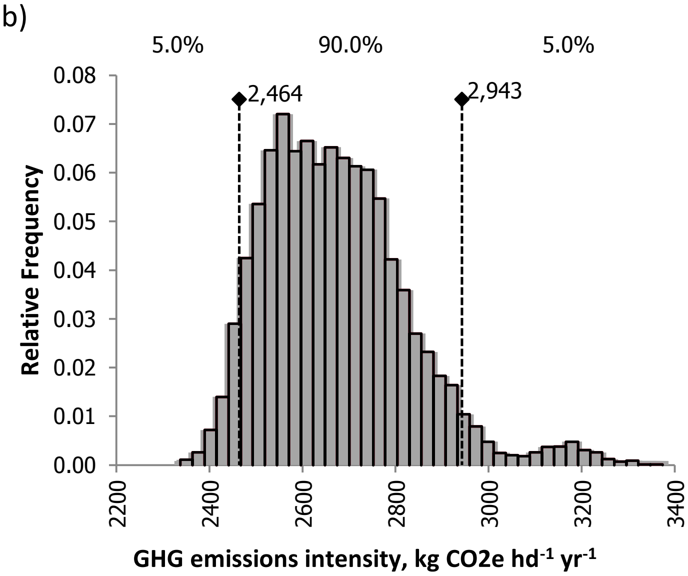
Boundaries	Calculated Value	Mean/Expected Simulation Value	Standard Deviation	5^{th} Percentile	95^{th} Percentile
Feedlot LCA	3119	3182	167	2950	3475
Feedlot LCA (no LUC)	2537	2673	156	2464	2943
Full LCA	16.55	16.67	0.319	16.23	17.23
Full LCA (no LUC)	11.97	12.35	0.315	11.91	12.90

Figure 7. Monte Carlo simulation probability distributions using @Risk: a) Feedlot LCA, b) Feedlot LCA (without LUC), c) Full LCA, d) Full LCA (without LUC). Designations for 5th and 95th percentiles shown.

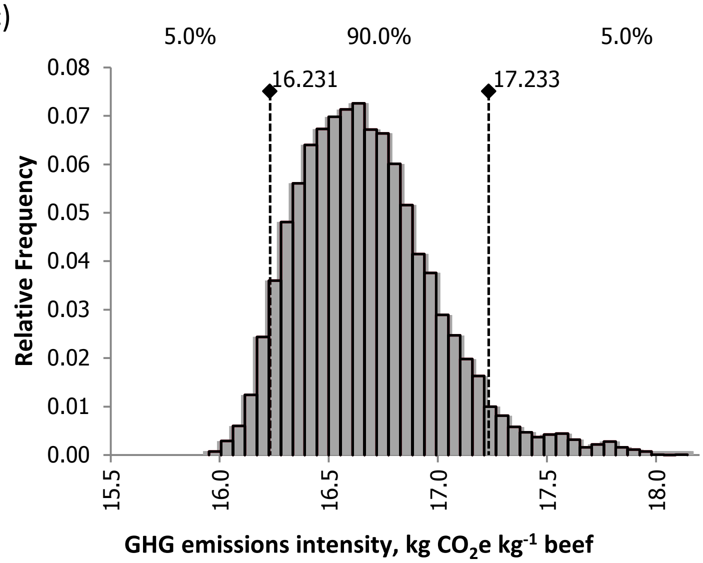
a)



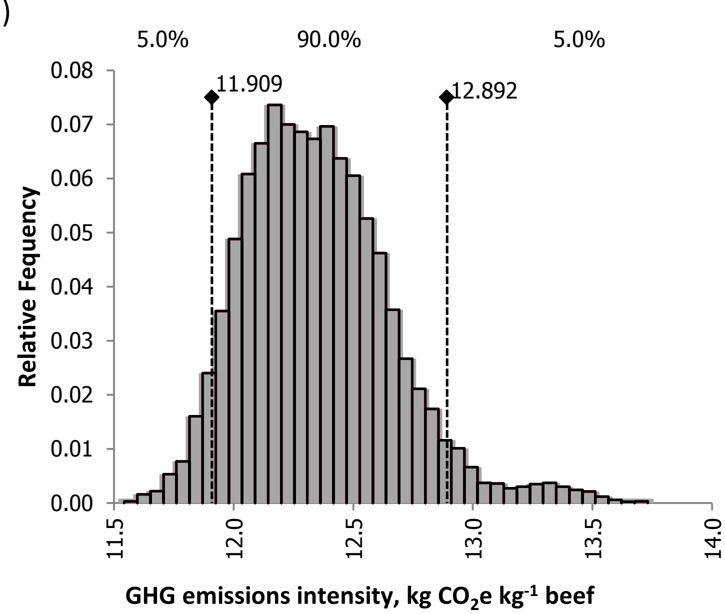
b)



c)


5.0%
90.0%
5.0%
16.231
17.233
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0.00
Relative Frequency
15.5
16.0
16.5
17.0
17.5
18.0
GHG emissions intensity, kg CO_2e kg^{-1} beef

d)


5.0%
90.0%
5.0%
11.909
12.892
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0.00
Relative Fequency
11.5
12.0
12.5
13.0
13.5
14.0
GHG emissions intensity, kg CO_2e kg^{-1} beef

3.7.1 Literature Comparison and the Importance of Boundary Definitions

Estimates of the life cycle emissions intensity of beef production from previous studies were summarized and compared with this analysis (Figure 8). The life cycle emissions intensity reported here is very similar to a recent assessment for beef cattle in the U.S. (using the same pasture intensity) (Pelletier et al., 2010), but the estimate provided here is roughly half of the intensity of two other estimates, at 25 and 32 $kgCO_2e\ kg^{-1}$ meat (Hamerschlag, 2011; Phetteplace et al., 2001). Previous estimates for U.S. beef were higher due to moisture and fat loss in cooking, plate loss, and spoilage (Hamerschlag, 2011), and higher enteric fermentation emissions and higher ill-defined N_2O emissions (Phetteplace et al., 2001). Life cycle emissions from other studies encompass Australia, North America, South America, Europe, and Asia and range from 12.7 $kgCO_2e\ kg^{-1}$ meat in U.S. (Pelletier et al., 2010) to 37.3 $kgCO_2e\ kg^{-1}$ meat in Brazil (Cederberg et al., 2009) (Figure 8).

Life cycle GHG emissions could be even greater where emissions are included from deforestation from the expansion of pasture (Cederberg et al., 2011). Previous global assessments of pasture and feed crops expansion have indicated these emissions could range from 38 to 53% of all emissions from livestock (Asner & Archer, 2010; H. Steinfeld et al., 2006). A recent study in Brazil found that deforestation associated with pasture expansion produced additional emissions in the range of 21.3 to 976 $kgCO_2e\ kg^{-1}$ meat, depending on whether these regional land use change emissions are allocated to all beef cattle in Brazil, or to the cattle only in areas of newly deforested land (Figure 8).

3.7.2 LCA Conclusions

Comparison of the LCA presented here with previous studies indicates that system boundaries have a defining impact on total GHG emissions from beef cattle. For all geographic areas, and where variable system boundaries are used, over a 1000-fold range in GHG emissions for beef cattle were identified (1.0 to 1013 $kgCO_2e\ kg^{-1}$). With only the variation of system boundaries in the U.S., nearly a 16-fold range in results was found from EPA Mandatory Reporting at 1.0 $kgCO_2e\ kg^{-1}$ beef to a LCA at 16.6 $kgCO_2e\ kg^{-1}$ beef. Even though shorter and more intensive feedlot finishing has been found to have lower GHG emissions compared to longer pasture finishing (Peters et al., 2010), much of the increase in LCA emissions from feedlot beef cattle found here was due to inclusion of pasture emissions associated with earlier stages of the animal 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 measurements of GHG emissions.

Figure 8. Comparison of estimated emissions from the life cycle of beef cattle production, including both feedlot and pasture. Data from Table 11. *LCA* (arrow) comprises manure management, enteric fermentation, off-site feed production emissions, including land use change, and pasture (Table 7), building on Figure 6. All studies include emissions from both feedlot and pasture, and where data was available, emissions were differentiated by color. Emissions of regional land use change from deforestation are allocated to beef production over three different areas (Cederberg et al. 2011): B) all of Brazil, A) the entire Legal Amazon Region, F) newly deforested land cleared for cattle. Error bars include 90% of Monte Carlo simulation values (Figure 7).

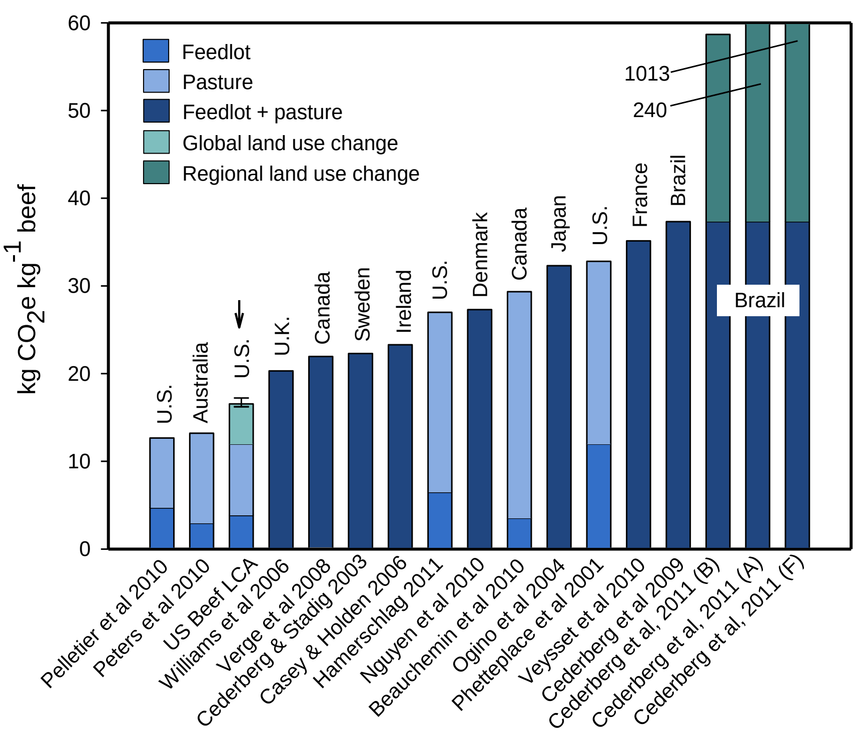


Table 9. Literature comparison of beef LCA results (kg CO_2e kg^{-1} beef).

Country	**Source**	**Feedlot Stage**	**Cow-Calf Stage**	**Land Use Change**	**Total**
USA	Pelletier et al. 2010	4.7	8.0	-	12.7[a]
Australia	Peters et al. 2010	2.9	10.3	-	13.2[b]
USA	Feedlot LCA	3.84[c]	8.13[d]	4.59	16.6[e]
U.K./Wales	Williams et al 2006				20.3[b]
Canada	Verge et al. 2008	-	-	-	21.9[f]
Sweden	Cederberg & Stadig 2003	-	-	-	22.3
Ireland	Casey &Holden 2006	-	-	-	23.3[f]
USA	Hamerschlag et al. 2011	6.5	20.5	-	27.0
EU/Denmark	Nguyen et al. 2010	-	-	-	27.3
Canada	Beauchemin et al. 2010	3.5	25.8	-	29.3[b]
Japan	Ogino et al. 2004	-	-	-	32.3
USA	Phetteplace et al. 2001	12.0	20.8	-	32.8[f]
France	Veysset et al. 2010	-	-	-	35.1[f]
Brazil	Cederberg, et al. 2009	-	-	-	37.3[b]
Brazil	Cederberg, et al. 2011[g]	-	-	21.3	58.7[b]
Brazil	Cederberg, et al. 2011[h]	-	-	202.7	240.0[b]
Brazil	Cederberg, et al. 2011[i]	-	-	976.0	1013.3[b]

[a] Conversion from kg CO_2e hd^{-1} to kg CO_2e kg^{-1} beef; assume 584 kg slaughter weight, 63% dressing percentage, and 75% meat:waste ratio

[b] Conversion from kg CO_2e kg^{-1} carcass weight to kg CO_2e kg^{-1} beef; assume 75% meat:waste ratio

[c] From Table 7

[d] adapted from Pelletier et al. 2010, 63% of life cycle emissions are from pasture

[e] Conversion from kg CO_2e hd^{-1} yr^{-1} o kg CO_2e kg^{-1} beef; assume 192 days on feed, 365 days yr^{-1}, 584 kg slaughter weight, 63% dressing percentage, and 75% meat:waste ratio

[f] conversion from kg CO_2e kg^{-1} live weight to kg CO_2e kg^{-1} beef; assume 63% dressing percentage, and 75% meat:waste ratio

[g] Rainforest land use change averaged over all of Brazil

[h] Rainforest land use change averaged over the Legal Amazon Region

[i] Rainforest land use change averaged over newly deforested land

Chapter 4: Economic Implications and Future Work

4.1 Implications for Regulatory Policy

Based on the analysis presented here, current federal regulation methodologies are incomplete measures of the actual life cycle emission of the feedlot system. If EPA were to choose to monitor only localized GHG emissions from beef cattle, the Annual Inventory appears to provide the better framework for direct GHG emission quantification relative to the limited Mandatory Reporting method. 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. These results demonstrate that a range of GHG emissions intensities can be determined by often arbitrary choices in system boundaries.

4.2.1 Environmental Economic Theory

Many have discussed the impact and costs of climate change on society. These costs (in 2005 dollars discounted at 3% over a 95 year period from 2010-2105) range from $10 trillion to $270 trillion with significant uncertainty associated with the estimation (Bosello, Carraro, & de Cian 2010). Many human activities emit greenhouse gases where the environmental cost is a negative externality not covered by the private cost of the activity. In welfare economics, this situation is a market failure and socially optimal activity will not be reached without some kind of market correction (Hanley, Shogren, &

White, 2007). Economists have long discussed means of correction; Ronald Coase has famously argued that defined property rights will provide incentive for polluters and pollution-affected individuals to come to a private, mutually beneficial economic transaction. This assumes that pollution "rights" are definitive and clear, something very difficult to do with regard to common pool resources like the atmosphere where quantification is often uncertain and impacts are spread globally. Alternatively, Arthur Pigou is credited with the idea of implementing a tax on the economic activity producing the externality equal to the marginal cost of damages from the activity (Hanley, Shogren, & White, 2007). Alternatively to this price-based approach, a quantity-based cap-and-trade system will theoretically achieve identical reduction in damages by distributing a limited number of emissions permits that can be traded and sold among emitters. There are many logistical and economic variables that must be considered by policy makers to produce the optimal results; these include tax and permit levels, whether permits are auctioned or given away, how market power is affected, correlation of the taxed entity to emissions level, uncertainty of damages, use of tax revenue, and incentive for innovation (Hanley, Shogren, & White, 2007).

4.2.2 Implications of Direct Carbon Pricing on the Economics of the Beef Industry

Though policy regimes do not yet regulate agricultural GHG emissions at this time, it is useful to know how future prices on carbon may affect food prices. Twomey and Webber use emissions intensities for nine primary fuel sources, as well as EPA and LCA methods to determine a 16.00 kg CO_2e kg^{-1} beef carbon intensity for beef which includes the following components: crop production, enteric fermentation, manure management, food manufacturing, food packaging, commercial food services, food retail facilities, residential food preparation, and transportation costs but does not include pasture or land use change emissions (Twomey & Webber, 2010). For the purposes of this analysis, the 16.6 kgCO_2e kg^{-1} kg beef found in the LCA presented above is used. Twomey and Webber suggest that the social cost of emitting CO_2e is in the range of $11 and $85 per tonne CO_2e. Thus, these direct carbon tax prices formed the upper and lower limits for this brief discussion. Using these parameters, the price of beef (if directly taxed based on the LCA) would increase by 9.9 to 53.5% (Table 12).

As the price of a product or good increases, the amount of people willing to pay these higher prices is reduced, resulting in a loss of demand that reduces the number of units that the industry produces. A review of 51 U.S.-based studies on price elasticity of demand for major food categories indicates mean price elasticity of 0.75 for beef with a range of 0.29-1.42 and 95% confidence interval of 0.67-0.83 (Andreyeva, Long, & Brownell, 2010). In other words, for every 10% increase in price, there will be a 7.5% decrease in demand. Demand elasticity is determined by a multitude of factors including availability of substitutes, household income, consumer preferences, expected duration of price change, and product's share of a household's income.

Applying this principle to the scenario modeled below, the U.S. beef industry as a whole will likely shrink by 2.5 to 18.9% as prices rise. This would result in a reduction of 2.7 to 21.0 Tg CO_2e emitted by the U.S. beef industry which is equivalent to 0.04 to 0.30% of net U.S. emissions. These results are extremely uncertain and highly dependent on demand elasticity as well many other unexplored parameters such as change in number of producers, change in production technology, supply elasticity, use and distribution of government tax revenue, and income elasticity of consumers. Also unclear is how beef substitutes (e.g. pork and chicken) would be taxed relative to beef and how the lower LCA of these meats would affect prices. Assuming limited structural change to the beef production, the \$32 billion year^{-1} beef industry would shrink by \$0.8 to 6.0 billion year^{-1}. However, it is likely that revenues would be returned to farmers in some form or some other tax/permit scheme used to limit the financial impact on beef producers.

Table 10. Carbon price and potential for reduced beef demand

	Units	Lower Price		Upper Price
Carbon price[a]	\$ ton^{-1} CO_2e	11	50	85
Baseline price for beef[b]	\$ kg^{-1} beef	5.59	5.59	5.59
Beef Carbon Intensity[c]	kgCO_2e kg^{-1}	16.6	16.6	16.6
Carbon Tax[d]	\$ kg^{-1}	0.18	0.83	1.41
New Beef Price	\$ kg^{-1}	5.77	6.42	7.00
Percent Increase	%	3.27%	14.86%	25.25%
Demand Decrease[e]	%	2.45%	11.14%	18.94%
Reduction of Beef Revenue to Industry	billion \$ yr^{-1}	-0.77	-3.51	-5.98
Reduction of Emissions	Tg CO_2e	-2.71	-8.81	-20.97
Percent of total US emissions reduced	%	-0.04%	-0.18%	-0.30%

[a] Carbon price range as estimated in (Twomey & Webber, 2010).
[b] Beef industry produced 26,291,800,000 lbs of beef in 2011 (USDA-NASS, 2012a). Assume average price is ~$120 per hundred pounds at end of 2011(USDA-NASS, 2012c). Conversion to kg yields average price of $2.64 kg^{-1} liveweight = $5.59 kg^{-1} beef assuming dressing percentage of 63% and 75% meat:waste ratio.
[c] Beef carbon intensity from this study; also closely correlated with Twomey & Webber, 2010 (though methodolgies are vastly different)
[d] carbon price * beef carbon intensity
[e] using a demand elasticity of 0.75 from Andreyeva, Long, & Brownell, 2010.

It is likely that any future carbon tax would be structured so as to tax large GHG sources as far "upstream" as possible (e.g. coal producers, oil refiners, etc) and allow the higher prices to permeate through the rest of the economy. Many of these industries are already taxed at lower levels and the convenience of merely increasing tax rates (as opposed to inventing a complex carbon trading network) is one of the attractive aspects of carbon taxation. LCA-driven carbon taxing for agriculture sectors would likely not occur until consensus LCA methods are formalized and more convenient GHG sources are subject to taxation.

While this analysis does not examine global impacts of beef prices, a recent examination of land use change explores potential future policy related to beef cattle (Dumortier et al. 2012). In this study, a 10% tax is imposed on U.S. fed steer prices. Modeled beef prices rise globally and consumption decreases. Specifically, U.S. production decreases by 17.06%; however, this is countered by increased production in Argentina (4.82%), Brazil (4.88%), Canada (6.69%), Indonesia (3.98%), and elsewhere since production in these areas is not subject to tax. However, resulting land use change in these countries will ultimately increase emissions by 37 to 85 kg CO_2e kg^{-1} U.S. beef reduced. This result argues 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. These findings emphasize how the attributional LCA approach used for this thesis may not be appropriate for informing the consequential effects of significant policy changes such as taxation and that more research is needed.

4.2.3 Logistics of Cap-and-Trade under Waxman-Markey (HR 2454) with regard to beef cattle

In the United States, there appears to be little momentum for legislative action on climate change and that most future climate policy will be determined by: 1) executive discretion of the EPA motivated by judicial mandate (*Mass v. EPA*, 2007), 2) international agreement and a top-down approach (Weiss & Jacobson, 1998), or 3) federal expansion of state-level policies and a bottom-up approach (Rabe, 2010). In discussing comprehensive federal approaches, the American Clean Energy and Security Act of 2009 (HR 2454), sponsored by Representatives Henry Waxman (D-CA) and Edward Markey (D-MA), was passed by the House of Representatives on June 26, 2009 and provides a model of what potential future legislative policy might be with regard to beef cattle. The bill included a cap-and-trade system that set a national limit on CO_2e emissions and granted a set number of emission allowances that could be traded and sold amongst regulated entities. The bill would have limited GHG emissions by 17% below 2005 levels by 2020 and by 83% by 2050.

The bill excluded the agricultural and forest industries from regulation and, instead, made them eligible for an incentive program titled “Offset Credit Program from Domestic Agricultural and Forestry Services”. This program, run by the USDA as opposed to the EPA, allowed entities to establish land management practices, manage carbon stocks, or improve waste management practices that could be proved to reduce, avoid, or sequester CO_2e. Though specific activities were never finalized, it appears that improvements in manure management could have made feedlot producers able to sell carbon offsets. Though enteric fermentation makes up a large portion of the cattle lifecycle (28% of the feedlot LCA presented here), it will not be regulated in this framework. Beef production might experience a small increase in costs as electric utilities and petroleum refineries will likely pass on their higher costs to consumers. However, feedlots will not be subject to specific caps on cattle transportation emissions since HR 2454 covers only “stationary” emission sources.

Due to the difficulty of delineating natural vs. anthropogenic emissions from agriculture and oppositional political interests, it appears that future comprehensive climate legislation will likely cover beef production emissions only as offsets and that the majority of emissions will go unregulated.

4.3 Future Work

The need for additional research is most apparent in evaluating the uncertainty associated with this life cycle assessment. In constructing the Monte Carlo simulation, only six parameters were associated with verifiable uncertainty. Thus, it is recommended that future empirical work be done to validate and assess the uncertainty of direct feedlot emissions (especially with regard to enteric fermentation and manure management). It appears that the vast majority of beef LCAs use some form of the IPCC emission factors in their inventories (Crosson et al., 2011); additional empirical evidence would be useful in constructing future life cycle assessment models. Additionally, few studies document the factors associated with assessment of pasture GHG emissions (though admittedly, these emissions are highly variable and difficult to quantify). However, additional work on measuring and modeling pasture emissions and potential mitigation efforts would be useful.

4.4 Conclusions

Policy frameworks for accounting for GHG emissions from livestock are currently in a relatively underdeveloped state. The quantitative methods underlying the EPA Mandatory Reporting rule are shown here to account for roughly 20% of the GHG emissions recognized by the more complete EPA Annual Inventory for beef cattle feedlots, with the latter approach giving relatively consistent emissions estimates despite variable input data (Figure 6). If the Annual Inventory is expanded to account for more GHG emissions associated with beef cattle feedlots, such as including emissions from crop production for cattle feed, the methods employed in the Mandatory Reporting rule would only account for roughly 8% of life cycle GHG emissions from feedlots. Despite being a relatively conservative approach to estimate emissions from beef cattle, the Mandatory Reporting rule will not be used by the EPA due to recent U.S. Congressional action to exclusively defund its implementation associated with livestock.

The EPA Annual Inventory was derived from IPCC methodology and this approach appears to be a standard for emissions in the LCAs of beef cattle feedlots (Crosson et al., 2011). It is clear from this analysis that when only accounting for localized emissions directly from the feedlot, other significant related GHG emissions that are attributed to the feedlot are not quantified, such as emissions from feed production. Inclusion of emissions from corn and alfalfa production raised Annual Inventory emissions by >60%. These environmental impacts from feed production are conventionally included in the inventory of attributional life cycle emissions from a feedlot. Emissions from other feed components, direct use of fossil fuels at the feedlot, and changes in soil carbon from spreading manure were found to be relatively minor contributions or savings in the feedlot life cycle.

In examining the economics of carbon and beef, it is clear that an increase in the price of beef (whether by a direct carbon tax, as in this analysis, or by some other means) has the potential to dramatically reduce beef consumption and GHG emissions associated with beef production by 2.7 to 21.0 Tg CO_2e. However, it appears that future comprehensive and state climate legislation will likely view beef production emissions only as offsets and that the majority of emissions will go unregulated.

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