Chapter 5
Quantifying Greenhouse Gas Sources and Sinks in Animal Production Systems

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Acronyms, Chemical Formulae, and Units

AA	Amino acids
AD	Anaerobic digestion
ADF	Acid detergent fiber
AGP	Antibiotic growth promoters
ASABE	American Society of Agricultural and Biological Engineers
B_0	Maximum methane production capacities
bLS	backward Lagrangian stochastic
BNR	Biological nitrogen removal
BW	Body weight
CH_4	Methane
CNCPS	Cornell Net Carbohydrate and Protein System
CO_2-eq	Carbon dioxide equivalents
CP	Crude protein
CSTR	Continuous stirred tank reactor
DDGS	Dried distillers grains with solubles
DE	Digestible energy
DFM	Direct fed microbials
DGS	Distillers grains with solubles
DIP	Dietary crude protein
DMI	Dry matter intake
DRC	Dry-rolled corn
EF	Emission factor
g	Grams
Gg	Gigagrams
GEI	Gross energy intake
GHG	Greenhouse gas
HCW	Hot carcass weight
HMC	High-moisture corn
IFSM	Integrated Farm System Model
kcal	Kilocalorie
kg	Kilograms
lb(s)	Pound(s)
LCA	Life cycle analysis
LU	Livestock unit
m	Meters
MCF	Methane conversion factor
ME	Metabolizable energy
mg	Milligram
MGA	Melengestrol acetate
MJ	Millijoules
NE	Net energy
N_{ex}	Nitrogen excreted
N	Nitrogen
N_2O	Nitrous oxide
NDF	Neutral detergent fiber
NFC	Non-fiber carbohydrate
NH_3	Ammonia
NPN	Non-protein nitrogen

NSP	Non-starch polysaccharide
O_2	Oxygen
OM	Organic matter
ppb	parts per billion
ppm	parts per million
RDP	Ruminal degradable protein
RFI	Residual feed intake
RMSPE	Residual mean square prediction error
SF_6	Sulfur hexafluoride
SFC	Steam-flaked corn
TAN	Total ammoniacal nitrogen
TDN	Total digestible nutrients
TKN	Total Kjeldahl nitrogen
TMR	Total mixed ration
UASB	Upflow anaerobic sludge blanket
UP	Unprocessed
U.S. EPA	U.S. Environmental Protection Agency
VFA	Volatile fatty acids
VS	Volatile solids
WDGS	Wet distillers grains with solubles
Ym	Methane conversion factor, percent of gross energy in feed converted to methane

5 Quantifying Greenhouse Gas Sources and Sinks in Animal Production Systems

This chapter provides guidance for reporting greenhouse gas (GHG) emissions associated with entity-level fluxes from animal production systems. In particular, it focuses on methods for estimating emissions from beef cattle (cow-calf, stocker, and feedlot systems), dairy cattle, sheep, swine, and poultry (layers, broilers, and turkey). Information provided is based on available data at the time of writing. In many cases systems are oversimplified because of limited data availability. It is expected that more data will become available over time. This chapter provides insight into the current state of the science and serves as a starting point for future assessments.

- Section 5.1 summarizes animal management practices and the resulting GHG emissions.
- Section 5.2 presents an overview of each production system and a general discussion of common management systems and practices.
- Section 5.3 describes the methods for estimating GHG emissions from enteric fermentation and housing (enteric fermentation being a much more significant emissions source than housing).
- Section 5.4 describes methods for estimating GHGs from manure management systems.
- Section 5.5 identifies research gaps that exist for quantifying GHGs from animal production systems. The intent of identifying research gaps is to highlight where improvements in knowledge can best improve the usefulness of this document at farm-, regional-, and industry-scales.

Ammonia Emissions in Animal Production Systems

Ammonia (NH_3), although not a GHG, is emitted in large quantities from animal housing and manure management systems and is an indirect precursor to nitrous oxide (N_2O) emissions as well as an environmental concern. Inside barns and housing units, NH_3 is considered an indoor air quality concern because it can have a negative impact on animal health and production. Volatilized ammonia can react with other compounds in the air to form particulate matter with a diameter of 2.5 microns. This fine particulate matter can penetrate into the lungs, causing respiratory and cardiovascular problems, and contribute to the formation of haze.

Information about ammonia has been included in this chapter and proposed quantification methods are presented in Appendix 5-C.

5.1 Overview

This section summarizes the key practices in animal management and the resulting GHG emissions that are discussed in detail in this chapter. The agricultural practices discussed include those required to breed and house livestock, including the management of resultant livestock waste. Emissions considered here include those from enteric fermentation (resulting from livestock digestive processes), livestock waste in housing areas, and livestock waste managed in systems (such as stockpiles, lagoons, digesters, solid separation, and others). Options for management changes that may result in changes in GHG emissions are also discussed.

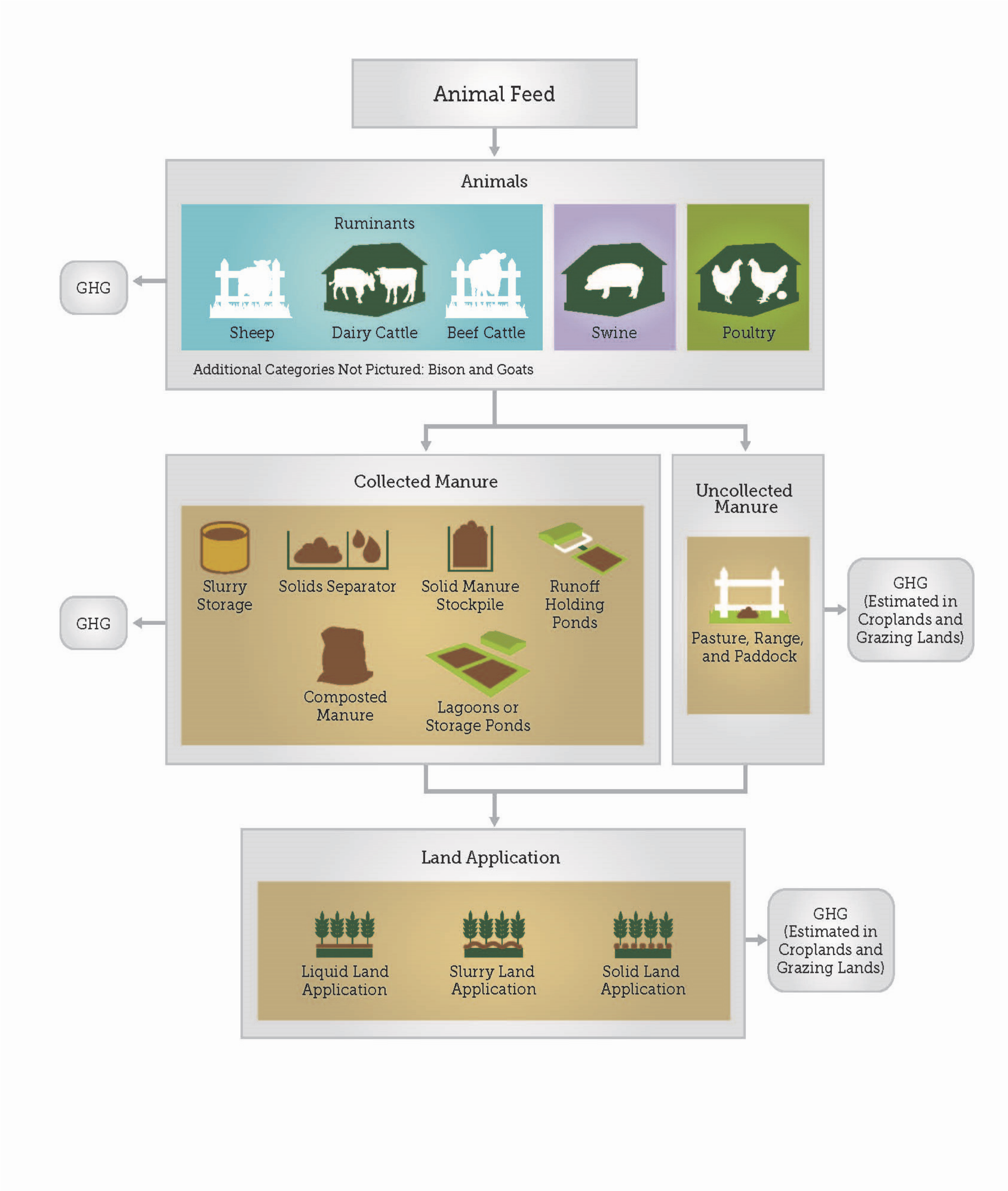
5.1.1 Overview of Management Practices and Resulting GHG Emissions

Animal production systems include agricultural practices that involve breeding and rearing livestock for meat, eggs, dairy, and other animal products such as leather, wool, fur, and industrial products like glue or oils. Farmers and other facility owners raise animals in either confined, semi-confinement, or unconfined spaces; the practices used to raise them are dependent on animal type, region, land availability, and individual preferences (e.g., conventional or "organic" standards). Regardless of the conditions in which animals are raised and housed, they produce GHG emissions. The magnitude of emissions depends primarily on the quality of the diet, the animals' requirements and intake (e.g., grazing, pregnant, lactating, performing work), and the types of systems in place to manage manure. The primary source of methane (CH_4) emissions from animal production systems is *enteric fermentation,* which is a result of bacterial fermentation during digestion of feed in ruminant animals. The second largest source of emissions from animal production systems is from the management of livestock manure. Methane emissions also occur from the digestive processes in monogastric animals; however, the quantity is significantly less than these other two sources. For simplicity, in the report, the term enteric fermentation refers to emissions from the digestive process of both ruminant and monogastric animals.

Manure management is the collection, storage, transfer, and treatment of animal urine and feces. Storage of animal manure has become increasingly popular as it allows synchronization of land application of manure nutrients with crop needs, reduces the need for purchased commercial fertilizer, and reduces potential for soil compaction due to poor timing of manure application. Depending on the storage and treatment practices, manure management has the added benefit of reducing air and water pollution. However, manure stored in anaerobic conditions results in the production and potential release of GHGs and odors. Greenhouse gas emissions from three solid manure storage/treatment practices (temporary stack and long-term stockpile, composting, and thermo-chemical conversion) and eight liquid manure storage/treatment practices (aerobic lagoon, anaerobic lagoon/runoff holding pond/storage tanks, anaerobic digestion, combined aerobic treatment system, sand-manure separation, nutrient removal, solid-liquid separation, and constructed wetland) are considered in the report.

Figure 5-1 provides an overview of the connections between feed, animals, manure, and GHG emissions in an animal production system. At the top of the conceptual model, livestock are fed a variety of diets. Ruminant animals eat feedstuffs and, through fermentation by the ruminal microbes, CH_4 is produced. Poultry and swine, although they do not release a significant amount of CH_4 through enteric fermentation, deposit manure into bedding, and upon manure decomposition, may release nitrous oxide (N_2O), CH_4 and ammonia (NH_3) into the atmosphere. Methodology to estimate emissions from bedding and dry manure in housing is similar to, and often parallel to, the method described for dry manure handling and storage systems. Manure from grazing livestock is left on fields or paddocks, and the manure may be collected to be treated and stored. Manure that has been collected and stored can be applied to croplands. GHG emissions from grazing lands and croplands are addressed in Chapter 3, Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems.

Figure 5-1: Connections Between Feed, Animals, Manure, and GHG for Animal Agriculture



5.1.1.1 Resultant GHG Emissions

For this report, methods are categorized according to those from enteric fermentation, housing, and manure management systems. The housing discussion includes emissions from manure deposited in the housing unit and manure that is managed inside those areas (such as interior stockpiles). Manure management includes emissions from managed, treated, and stored manure.[1]

Enteric Fermentation and Housing Emissions

Methane-producing microorganisms, called methanogens, exist in the gastrointestinal tract of many animals. However, the volume of CH_4 emitted by ruminants is vastly different from that of other animals because of the presence and fermentative capacity of the rumen. In the rumen, CH_4 formation is a disposal mechanism by which excess hydrogen from the anaerobic fermentation of dietary carbohydrate can be released. Control of hydrogen ions through methanogenesis assists in maintenance of efficient microbial fermentation by reducing the partial pressure of hydrogen to levels that allow normal functioning of microbial energy transfer enzymes (Morgavi et al., 2010). The only GHG of concern resulting from enteric fermentation is CH_4. Respiration chambers equipped with N_2O analyzers indicate that enteric fermentation does not result in the production of N_2O (Reynolds et al., 2010). Methane can also arise from hindgut fermentation, but the levels associated with hindgut fermentation are much lower than those of foregut fermentation.

Background: Ruminants

Ruminants are animals that have four-chambered stomachs, which allow for easier digestion of high-fiber, hard-to-digest feedstuffs. They include:

- Cattle
- Goats
- Sheep
- Deer
- American Bison

Because the magnitude of enteric emissions is so great and, therefore, a significant contributor to many countries' GHG emissions, decades of research have gone into characterizing, understanding, and attempting to mitigate enteric CH_4 emissions. A fundamental challenge in this type of research has been the measurement of these emissions.

Methane, N_2O, carbon dioxide (CO_2), and NH_3 are produced from livestock feces and urine, and some gaseous forms are emitted soon after manure excretion. In dry-lot situations, feces and urine are deposited on the pen surface and are mixed via animal hoof action. Microorganisms in the feces or underlying soil metabolize nutrients in the manure to produce GHGs. In feedlots, where manure is normally cleaned from pens once or twice per year, distinctive, hard-packed layers of manure and soil may develop that produce microenvironments favorable to oxidative and reductive processes (Woodbury et al., 2001; Cole et al., 2009b). Periods of rainfall or dry conditions may alter the microbial and chemical nature of the pen surface. Production of CH_4 and N_2O occur in the underlying manure/soil layers and in water-saturated areas where oxygen is limited, such as wet areas of the pen around water troughs and depressions that collect rain water and snow melt. In contrast, most NH_3 produced in the pen probably comes from fresh urine spots on the pen surface. To date, few measurements of GHG emissions from feedlot or dry-lot pen surfaces have been made.

Runoff from dry-lot and feedlot pens is normally collected in retention ponds (more typical in feedlots), or lagoons (more common in dairies). In some cases, runoff may undergo partial removal of suspended solids in settling basins (feedlots and dairies) or in mechanical separators (dairies only) that parallels treatment of manure collected in these same systems. Losses of GHGs and NH_3

[1] Emissions from manure deposited on grazing lands are addressed in Chapter 3: Croplands and Grazing Lands.

from these facilities depend upon climatic factors and the oxidative-reductive potential, pH, and chemistry of the effluent in the pond or lagoon. A limited number of studies have measured GHG or NH_3 emissions from retention ponds or lagoons.

Manure Management

Manure is managed in a wide variety of systems. The resulting GHG emissions differ by GHG and magnitude of emissions per quantity of manure. Table 5-1 provides an overview of the liquid and solid manure systems considered in this report and the resulting GHGs.

Table 5-1: Overview of Manure Management Systems and Associated Greenhouse Gases

Storage and Treatment Practices		Estimation Method			Description
		CH_4	N_2O	NH_3 [a]	
Solid Manure	Temporary and long-term storage	✓	✓	✓	Manure may be stored temporarily for a few weeks to avoid land application during unfavorable weather or it can be stored for several months.
Solid Manure	Composting	✓	✓	✓	Composting involves the controlled aerobic decomposition of organic material and can occur in different forms. Estimation methods are provided for in vessel, static pile, intensive windrow, and passive windrow composting.
Solid Manure	Thermo-chemical conversion				Thermo-chemical conversion involves the combustion of animal waste, converting CH_4 to CO_2. Pyrolysis/gasification is one method that has received much interest. No method is provided as GHGs are considered negligible.
Liquid Manure	Aerobic lagoon	✓	✓	✓	Aerobic lagoons involve the biological oxidation of manure as a liquid with natural or forced aeration.
Liquid Manure	Anaerobic lagoon/runoff holding ponds/storage tanks	✓	✓	✓	Anaerobic lagoons are earthen basins that provide an environment for anaerobic digestion and storage of animal waste. Lagoons may be covered or uncovered and have a crust or no crust formation. Runoff and holding ponds are constructed to capture and store runoff from feedlots and dry-lots. In some cases wash water from dairy parlors may be stored in holding ponds. Storage tanks typically store slurry or wastewater that was scraped or pumped from housing systems.
Liquid Manure	Combined aerobic treatment system	✓	✓	✓	This process involves removing solids using flocculation and then composting the solid stream and aerating the liquid stream of manure.
Liquid Manure	Anaerobic digester	✓			Anaerobic digesters are manure treatment systems designed to maximize conversion of organic wastes into biogas. These can range from covered anaerobic lagoons to highly engineered systems. Methane gas leakage is the main source of GHG emissions; NH_3 and N_2O leakage is negligible.
Liquid Manure	Sand–manure separation				Manure is separated from sand and bedding by mechanical and sedimentation separation. No method is provided as emissions are negligible. Separated liquids and solids could be inputs into other storage systems.
Liquid Manure	Nutrient removal				There are four main nitrogen removal approaches: biological nitrogen removal, Anammox (i.e., anaerobic ammonium oxidation), NH_3 stripping, ion exchange, and struvite crystallization. No method is provided due to limited quantitative information on GHG generation from nutrient removal systems.

Storage and Treatment Practices		Estimation Method			Description
		CH_4	N_2O	NH_3 [a]	
	Solid–liquid separation				Mechanical separation of liquids and solids through screens, centrifuges, pressing, filtration, or microscreening. Separated liquids and solids could be inputs into other storage systems.
	Constructed wetland				Typically consist of wetland plants growing in a bed of highly porous media. No method is provided as emissions are negligible; GHG sinks are noted to likely be greater than emissions.

[a] Although NH_3 is considered in this chapter as an important precursor to particulate formulation (affecting radiation balance) and GHGs and is a key element of discussion, NH_3 itself is not a GHG. Therefore, methods for estimating NH_3 emissions are provided in Appendix 5-C.

An entity can reduce its GHG emissions from manure by utilizing alternative treatment options and/or management systems. Anaerobic digesters do not reduce the amount of CH_4 released but offer an option to capture and convert the CH_4 to CO_2 and energy through combustion. Digesters offer both CH_4 reductions as well as GHG avoidance by reducing an entity's electricity demand.

Combined Aerobic Treatment Compared to Anaerobic Lagoons

A combined aerobic treatment system involves the treatment of a manure stream with flocculants to remove the majority of solids from the stream. The solids portion is composted while the remaining liquid is transferred to a storage tank where it is aerated. Methane is avoided by aerobically treating the solids via composting while NH_3 in the wastewater is avoided via nitrification. The GHGs resulting from a combined aerobic treatment are only 10 percent of what would be emitted from an anaerobic lagoon, thus combined aerobic treatments represent a potential mitigation option for entities.

5.1.1.2 Management Interactions

Table 5-2 depicts the key types of information desired for estimating GHG emissions from an animal production facility. This table illustrates the attributes of a system that have the greatest influence over emissions within each component. A number of existing models can be used to estimate GHG emissions that utilize the key activity data indicated in Table 5-2.

Table 5-2: Desired Activity and Ancillary Data for Estimating GHG Emissions from Animal Production Systems

General Category	Specific Data	Cattle				Sheep	Swine	Poultry	Goats	Amer. Bison
		Cow-calf	Stockers	Feedlot	Dairy					
Animal Characteristics	Body weight	•	•	•	•	•	•	•	•	•
	Body condition score	•	•		•	•				
	Stage of production (dry, lactating, pregnant)	•			•	•				
Dietary Facto	Diet intake (or factors that can be used to predict intake)	•	•	•	•	•	•	•	•	•

General Category	Specific Data	Cattle				Sheep	Swine	Poultry	Goats	Amer. Bison
		Cow-calf	Stockers	Feedlot	Dairy					
	Type of forage (conserved or grazed, pasture composition, stage of plant growth)	•	•		•	•			•	•
	Diet dry matter intake, crude protein, neutral detergent fiber, acid detergent fiber, non-structural carbohydrates, fiber, fat, energy content	•	•	•	•	•	•	•	•	•
	Diet digestibility and/or rate of passage	•	•	•	•	•	•			
	Degradability of carbohydrates and proteins	•		•	•					
	Supplementation practices – type (e.g., grains, protein, liquid, dry blocks, non-protein nitrogen) and quantity	•	•			•			•	•
	Supplemental or diet ionophore concentration	•	•	•	•					
	Dietary beta-agonists			•			•			
Nutrient Excretion: Quantity	Carbon, nitrogen, and volatile solids	•	•	•	•	•	•	•	•	•
Other Animal Factors	Growth promoting implants		•	•						
Manure Management Factors	Animal management regimen used to spread manure over pasture to reduce concentration near water or feed sources	•	•		•	•			•	•
	Soil type	•	•	•	•	•	•	•	•	•
	Practices to control runoff from pastures/lots/fields	•	•	•	•	•	•	•	•	•
	If housed, the length of time they are housed, animal concentration, manure handling procedures	•	•	•	•	•	•	•	•	•
	Type of manure collection/storage system			•	•		•	•	•	
	Frequency of manure collections and composition			•	•		•	•	•	
	Bedding/litter use and source			•	•		•	•	•	

5.1.2 System Boundaries and Temporal Scale

System boundaries are defined by the coverage, extent, and resolution of the estimation methods. The methods in this report can be used to estimate GHG emission sources that occur within the production area of an animal production system, including the animals, animal housing, and manure handling, treatment, and storage. Methane emissions from enteric fermentation, as well as the CH_4 and N_2O emissions from manure management systems or manure stored in housing, are considered in this report. Ammonia, while not a GHG, is a precursor to N_2O formation and is, therefore, included, primarily in Appendix 5-C. The act of transporting manure to the field for land application is included in the production area boundary, but emissions from vehicle transport are not included in the scope of this report as there are many variables that would determine emissions from vehicles (age of vehicle, type, fuel efficiency, idle time), and they are not direct agricultural emissions and could instead be considered part of the transport sector (off-road). Additionally, this report does not encompass a full life cycle analysis (LCA) of GHG emissions from animal production systems. The adjacent text box summarizes several studies on LCAs for animal production systems; however, they are not utilized in this report. Emissions that result following manure application are addressed separately in Chapter 3, Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems.

Qualitative Discussion on Manure Sources

Estimation methods are not available for some sources. Qualitative discussion is provided for:

- Sand-Manure Separation
- Nutrient Removal
- Solid-Liquid Separation
- Constructed Wetlands
- Thermo-chemical Conversion

For emissions from animal production systems, the methods provided have a resolution of individual herds within an entity's operation. A herd is defined as a group of animals that are the same species, graze on the same parcel of land (same diet composition), and utilize the same manure management systems. Emissions are estimated for each individual herd within an operation and then added together to estimate the total animal production emissions for an entity. The animal production totals are then combined with emissions from croplands, grazing lands, and forestry to determine the overall emissions from the operation based on the methods provided in this document. Emissions are estimated on an annual basis.

5.1.3 Summary of Selected Methods/Models/Sources of Data

The Intergovernmental Panel on Climate Change (IPCC, 2006) has developed a system of methodological tiers related to the complexity of different approaches for estimating GHG emissions. Tier 1 represents the simplest methods, using default equations and emission factors provided in the IPCC guidance. Tier 2 uses default methods, but emission factors that are specific to different regions. Tier 3 uses country-specific estimation methods, such as a process-based model. The methods provided in this report range from the simple Tier 1 approaches to the most complex Tier 3 approaches. Higher-tier methods are expected to reduce uncertainties in the emission estimates, if sufficient activity data and testing are available.

Estimating CH_4 emissions from enteric fermentation in swine, goats, American bison, llamas, alpacas, and managed wildlife use Tier 1 methods. Enteric emissions from sheep are estimated using the Howden equation (Howden et al., 1994), and emissions from dairy production systems are estimated using the Mitscherlich 3 (Mits3) equation (Mills et al., 2003) as provided in the Dairy Gas Emissions Model (DairyGEM) (Rotz et al., 2011a). Emissions from beef cows are estimated using the IPCC Tier 2 approach. Emissions from feedlots are estimated using a modification of the IPCC Tier 2 approach.

Life Cycle Analysis of Cattle Production Systems

Peters et al. (2010) reported that the estimated carbon footprint of cattle production systems around the world ranged from 8.4 kg of CO_2-eq (kg HCW)$^{-1}$ (HCW=hot carcass weight) in an African pastoral system to 25.5 kg CO_2-eq (kg HCW)$^{-1}$ in an intensive Japanese grain feeding system. Five North American studies (Verge et al. (2008) and Beauchemin et al. (Sweeten, 2004; 2010) in Canada, Pelletier et al. (2010) and Lupo et al. (2013) in the U.S. Midwest, and Stackhouse et al. (2012) and Stackhouse-Lawson et al. (2012) in California) estimated the carbon footprint of various beef cattle production systems: The carbon footprint for the total beef production systems ranged from 10.4 to 19.2 kg CO_2-eq (kg final body weight) $^{-1}$ (or 16.7 to 32.5 kg CO_2-eq (kg HCW)$^{-1}$). Sixty four to 80 percent of the total CO_2-eq was produced in the cow-calf sector of production; whereas 8 to 20 percent of CO_2-eq was produced in the stocker phase, and only 12 to 16 percent was produced during the finishing phase. The majority (55 to 63 percent) of the total CO_2-eq was enteric CH_4, 18 to 23 percent was manure N_2O, and 14 to 24 percent was from fossil energy use and secondary emissions.

In general, the daily carbon footprint was greater during the grazing (stocker) phase than during the feedlot finishing phase. Both Pelletier et al. (2010) and Stackhouse et al. (2012) reported that the carbon footprint was slightly lower for calves that were weaned and went directly to the feedlot (21.1 and 23.0 kg CO_2-eq (kg HCW)$^{-1}$ or 2,382 and 3,493 kg head^{-1}, respectively) than for cattle that went through a stocker grazing phase before entering the feedlot (22.6 and 26.1 kg CO_2-eq (kg HCW)$^{-1}$ or 2,904 and 4,522 kg CO_2-eq head^{-1}, respectively). Pelletier et al. (2010) and Lupo et al. (2013) both reported that the carbon footprint of grass-finished cattle was greater than for calves that were weaned and went directly to the feedlot. These differences are due in part to slower weight gain and lighter final body weights and carcass weights of grass-fed cattle than cattle finished on grain- and byproduct-based diets in the feedlot.

Most LCAs assume that carbon sequestration is minimal in established, unfertilized pastures. Phetteplace et al. (2001) and Liebig et al. (2010) suggested there may be some small net carbon sequestration, in established native pastures. However, Liebig et al. (2010) noted that fertilized, improved pastures had net CO_2-eq emissions; primarily because of increased losses of N_2O from fertilizer nitrogen. Lupo et al. (2013) noted that the assumed carbon sequestration of pastures (equilibrium vs. net sequestration) affected the carbon footprint of grass-finished cattle; however, regardless of the carbon sequestration assumption, grass-finished cattle had a greater carbon footprint than grain-finished cattle.

For manure management, the IPCC Tier 2 methodology is used for CH_4 emissions from temporary stack and long-term stockpile, CH_4 and N_2O emissions from composting, and N_2O emissions from aerobic lagoons. The Sommer model is used to estimate CH_4 emissions from anaerobic lagoons.

All methods include a range of data sources from operation-specific data to national datasets. Operation-specific data will need to be collected by the entity and generally are activity data related to the farm and livestock management practices (e.g., dietary information, volatile solids content of manure). National datasets are recommended for ancillary data requirements, such as climate data and soil characteristics.

A summary of proposed methods and models for estimating GHG emissions from animal production systems is provided in Table 5-3.

Table 5-3: Summary of Sources and Proposed GHG Estimation Methods for Animal Production Systems

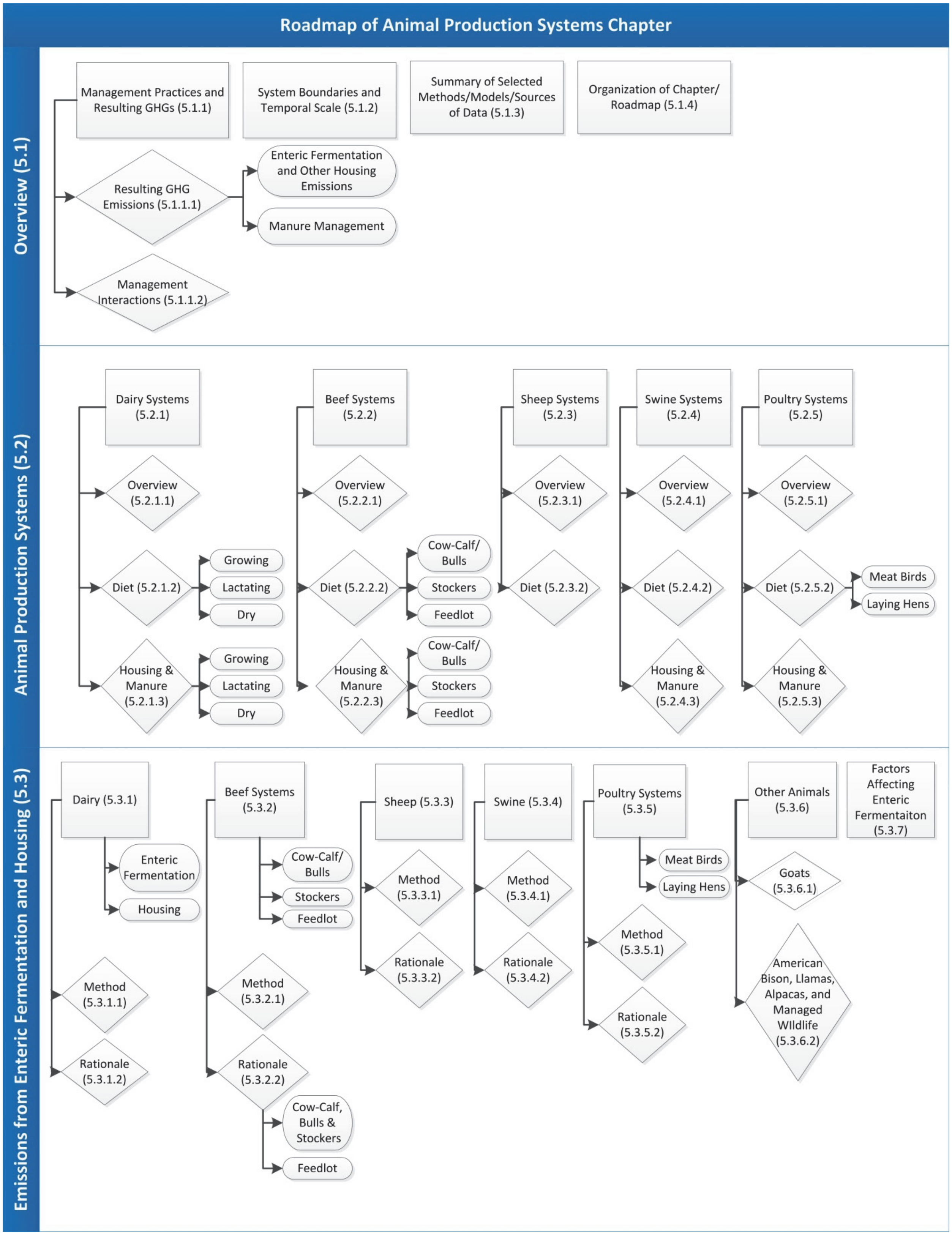
Section	Source	Method
Animal Production Systems, Including Enteric Fermentation and Housing Emissions		
5.3.1.2	Dairy Cattle	Mits3 equation; ASABE Standard D384.2 and IPCC Tier 2 (housing)
5.3.2.2	Beef Cattle	Modified IPCC Tier 2 (enteric and housing); ASABE Standard D384.2 (housing)
5.3.3.2	Sheep	Howden equation for grazing sheep (Howden et al., 1994) and Blaxter and Clapperton (1965) for feedlot sheep
5.3.4.2	Swine	IPCC Tier 1 (enteric methane); ASABE Standard D384.2 and IPCC Tier 2 (housing)
5.3.5.2	Poultry	IPCC Tier 1; ASABE Standard D384.2 and IPCC Tier 2 (housing)
5.3.6.1	Goats	IPCC Tier 1
5.3.6.2	American Bison, Llamas, Alpacas, and Managed Wildlife	IPCC Tier 1
Manure Storage and Treatment		
Temporary Stack & Long-Term Stockpile		
5.4.1.2	Methane	IPCC Tier 2 using U.S. EPA Inventory emission factors (EFs) and diet characterization
5.4.1.4	Nitrous Oxide	IPCC Tier 2 using U.S.-based EFs and monthly data
Composting		
5.4.2.2	Methane	IPCC Tier 2 with monthly data
5.4.2.4	Nitrous Oxide	IPCC Tier 2
Aerobic Lagoon		
5.4.3.2	Methane	Methane Conversion Factor for aerobic treatment is negligible and was designated as 0% in accordance with IPCC
5.4.3.4	Nitrous Oxide	IPCC Tier 2 using IPCC EFs
Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks		
5.4.4.2	Methane	Sommer model based on fractions of volatile solids (Møller et al., 2004)
5.4.4.4	Nitrous Oxide	Function of the exposed surface area and U.S.-based emission factors
Anaerobic Digestion		
5.4.5.2	Methane	IPCC Tier 2 using Clean Development Mechanism EFs for digester types to estimate CH_4 leakage from digesters
Combined Aerobic Treatment Systems		
5.4.6.2 5.4.6.2	Methane Nitrous Oxide	10% of emissions from estimation of liquid manure storage and treatment – anaerobic lagoon, runoff holding pond, storage tanks
Other Treatment Methods		
5.4.7	Sand–Manure Separation	No method provided because GHG emissions are negligible
5.4.8	Nutrient Removal	Not estimated due to limited quantitative information
5.4.9	Solid Liquid Separation	No method provided because GHG emissions are negligible
5.4.10	Constructed Wetland	No method provided because emissions are negligible; GHG sinks are noted to likely be greater than emissions
5.4.11	Thermo-chemical Conversion	No method provided as GHG emissions are negligible

5.1.4 Organization of Chapter/Roadmap

The remainder of this chapter is organized into four primary sections, as illustrated in Figure 5-2. Section 5.2 provides overviews of dairy cattle, beef cattle, sheep, swine, and poultry production systems and provides information on diet and housing. Section 5.3 provides the methods for estimating GHGs from housing, primarily focusing on GHGs from enteric fermentation. Methods are also provided for all the species described in Section 5.2, plus additional animal types (i.e., goats, American bison, llamas, alpacas, and managed wildlife). Section 5.4 provides the methodology for estimating emissions from different manure management systems. Methodology is provided to estimate CH_4 and N_2O from temporary stack and long-term stockpiles, composting, aerobic lagoons, anaerobic lagoons, and combined aerobic treatment systems. Section 5.4 also provides methods for estimating CH_4 from anaerobic digestion. A qualitative discussion is provided for sand-manure separation, nutrient removal, solid-liquid separation, constructed wetlands, or thermo-chemical conversion. Section 5.5 presents research gaps for both enteric fermentation and manure management.

There are five appendices to the animal production systems chapter of this report. Appendix 5-A provides Ym adjustment factors for calculating enteric CH_4 from feedlot cattle. Appendix 5-B provides nutritional information about animal feedstuffs (Ewan, 1989; Preston, 2013). Appendix 5-C discusses available methodologies for estimating NH_3 emissions from animal production systems. Appendix 5-D describes the shape factors and related equations that can be applied in Appendix 5-C to more accurately estimate emissions from manure stockpiles that are shaped differently (as surface area partially determines emissions). Appendix 5-E provides a detailed review of models evaluated for suitability for estimating emissions from animal production systems.

Figure 5-2: Animal Production Systems Road Map



Appendix 5-A: Enteric CH_4 from Feedlot Cattle – Methane Conversion Factor (Ym)

Appendix 5-B: Feed Stuffs Composition Table

Appendix 5-C: Estimation Methods for Ammonia Emissions from Manure Management Systems

Appendix 5-D: Manure Management Systems Shape Factors

Appendix 5-E: Review of Enteric Fermentation Models

5.2 Animal Production Systems

This section provides discussion on the production systems for beef and dairy cattle, sheep, swine, and poultry. This provides the background necessary for understanding Section 5.3, which covers GHG emissions from animal production systems.

5.2.1 Dairy Production Systems

5.2.1.1 Overview of Dairy Production Systems

The U.S. dairy production system is comprised of several key processes for dairy cattle, their manure, and their end products (meat, dairy) as depicted in Figure 5-3. This conceptual model provides an overview of the typical dairy system, following cattle from birth to slaughter and following manure from the animal through a management system. Manure is produced during each stage, and depending on the location, is managed differently. The management of the resultant manure has implications on the quantity of GHG emissions and sinks; the key practices are discussed in detail below. The estimation methods in this chapter include discussions for emissions from enteric fermentation, housing, and manure management and are not a full LCA.

The U.S. dairy industry is composed primarily of four major segments of production: 1) calf rearing; 2) replacement heifers; 3) lactating cows; and 4) nonlactating (dry) cows. The U.S. dairy cattle population in 2012 consisted of approximately 9.2 million milk cows and first calf heifers and approximately 4.6 million replacement heifers. The majority of dairy cattle in the United States are Holstein (Holstein-Friesian), followed by Jersey, with smaller numbers of Guernsey, Brown Swiss, and Ayrshire. Over the last 65 years there have been dramatic increases in milk production per animal, due to changes in herd management, nutrition, composition, and breeding programs. Present-day dairy herds are dominated by Holstein cows (90 percent) as opposed to a mix of the five most common breeds (Jersey, Guernsey, Ayrshire, Brown Swiss, and Holstein) as was common in the 1940s. With a change in breed dominance and enhanced genetics, the typical milk production per cow has increased from 2,074 to 9,193 kg of milk per year (Capper et al., 2009).

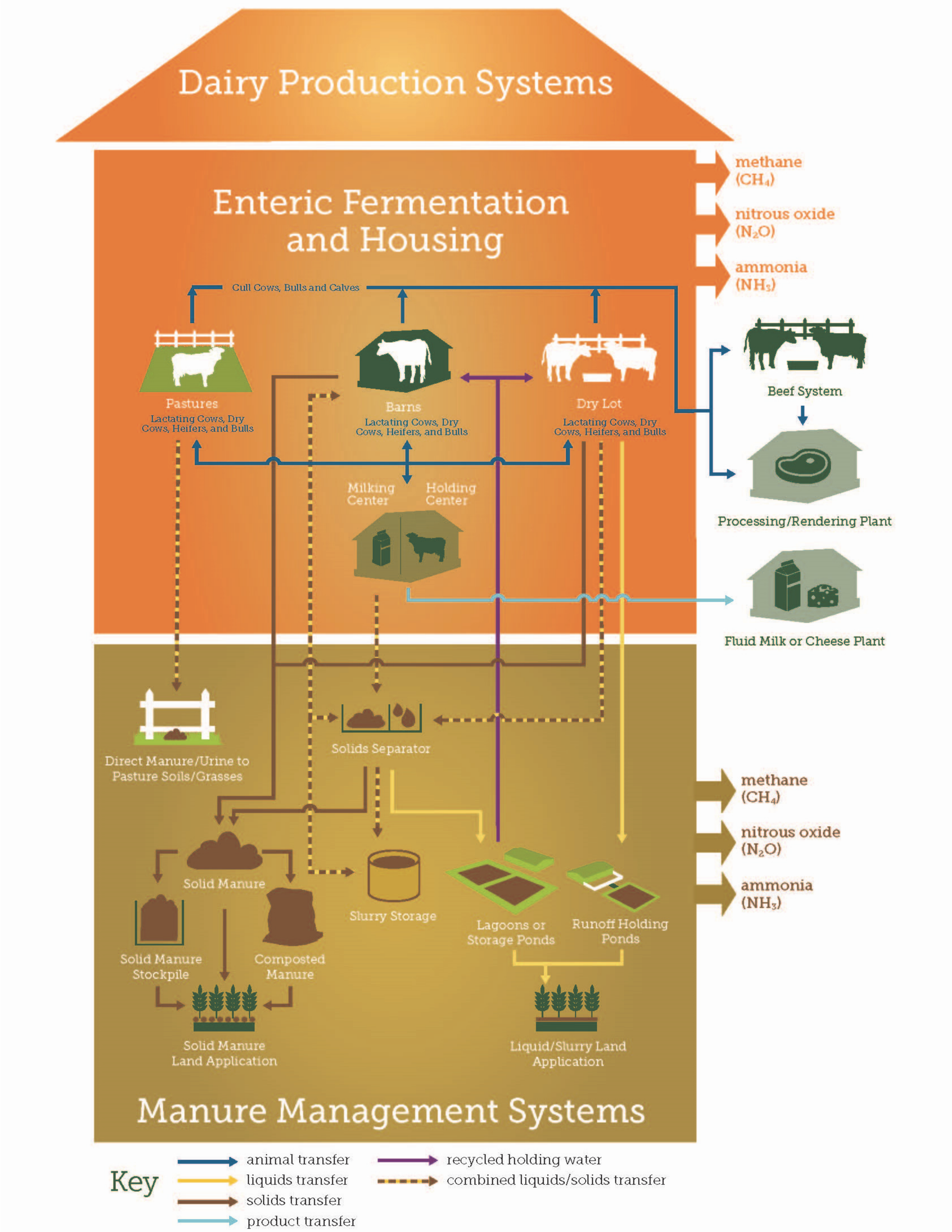
5.2.1.2 Diets for Dairy Cattle

Cows in intensive dairy production systems are fed diets that reflect regionally available feeds and typically contain between 40 and 60 percent concentrates, such as feed grains, protein supplements, and byproducts such as distiller's grains. Typical diets include corn silage, alfalfa or grass silage, alfalfa hay, ground or high-moisture shelled corn, soybean meal, fuzzy whole cottonseed, and often byproduct feeds (e.g., corn gluten, distiller's grains, soybean hulls, citrus pulp, beet pulp). Byproduct feeds may make up a large portion of the diet composition, providing key nutrients and a means of disposal for otherwise landfilled ingredients. Proximity to crop processing plants and industries may dictate the availability of byproduct feeds by region.

Growing Heifers

Diets for growing heifers are formulated based on growth rate and stage of rumen development. Diets range from liquid diets (e.g., milk or milk replacer) in newborn calves to pelleted complete feeds in the growing calf (e.g., calf starter) to diets that are similar to that offered to lactating cows as the cows grow and rumens develop. Roughage content of the diet increases as the rumen develops with hay or silage often offered in conjunction with a calf starter during a transition period. Following that transition, typical feeds include those listed above. Feeds are often mixed together in a mixer and fed as a Total Mixed Ration (TMR). In some cases, feed not consumed by the lactating herd is fed to growing heifers when the rumen is fully developed (> 9 months of age).

Figure 5-3 Conceptual Model of Dairy Systems in the United States



Lactating Cows

Diets for lactating cows are formulated by target milk production or stage of lactation, which reflects the differences in energy and protein required for different amounts of milk produced. Peak lactation occurs about 60 days after calving, and production slowly declines over the next several months. Feedstuffs are commonly blended together in a mixer and fed as a TMR.

Dry Cows

Dry cow diets are often formulated into two stages: far-off dry and close-up dry. During the far-off dry period, cows are fed diets with high forage content (>60%) using ingredients similar to that fed to the lactating herd. As dry cows approach calving, energy content of the diet increases by decreasing forage to include more concentrate feeds and mineral formulation changes in order to avoid pre- and post-partum metabolic disorders that often center around calcium mobilization as the cow begins to lactate. Feedstuffs are commonly blended together in a mixer and fed as a TMR

5.2.1.3 Dairy Housing and Manure Handling

Two general dairy farm types can be distinguished in the United States: confinement feeding systems (including barns and dry-lots) and pasture-based systems (USDA, 2004a). Typical housing systems for confinement feeding operations include tie stall barns, freestall barns, freestall barns with drylot access, and drylots. Drylot systems consist of housing animals in pens similar to beef cattle feedlots, but at a lower stocking density. In pasture-based systems, cattle graze pasture for periods of time, based on feed availability and environmental conditions, and are housed in barns and fed stored feed when pasture is not available. The dairy cattle lifecycle production phase is generally divided into three segments: growing animals (calves and replacement heifers), lactating mature cows, and dry mature cows. Nutrient needs, and therefore diets, and intake are very different between the different lifecycle phases: growing cattle (calves and heifers), lactating cows, and dry cows. Housing and manure management systems vary considerably throughout the country and can differ in a region and by the size of the herd. In cases where housing and manure management varies by animal group (e.g., heifers, dry, and lactating cows), estimates of GHG emissions from one group are not applicable to other groups. When housing and manure management are similar between groups (e.g., all cattle on dry-lots), diet and intake adjustment factors can be used to compare GHG emissions for the different groups.

With the exception of calves, replacement heifers and dry cows may be housed and managed in similar ways as lactating cows. When this is the case, much of the discussion is relevant to the three groups. In cases where the lactating herd is managed in confinement but replacement and dry animals are managed on pasture or in dry-lots, emissions from lactating cattle are not applicable not only due to differences in diet and intake but also due to housing differences. There are no readily available studies that have focused strictly on emissions from dairy calf management and housing. Summarized below are key characteristics of difference in housing by life cycle phase of a dairy cow.

- **Growing (calves and replacement heifers).** Following birth, calves are usually removed from the cow within a few hours and are typically reared on milk or milk replacer in calf hutches or barns for three to seven weeks until weaning. Female calves (replacement heifers) are typically moved to group housing (e.g., super hutches, transition barns, open housing, or pasture) until they reach appropriate breeding weight at about 14 to 15 months of age. Some replacements are contract-reared by heifer growers or sold. Following breeding, heifers are often raised in lots, pasture, or barns until they are ready to calve. Manure in group housing may be handled as a solid (bedded pack or compost barn) or as a slurry, similar to that described below for lactating cows in freestall barns.

- **Lactating Cows.** Heifers typically have their first calf at about 23 to 24 months of age, after which they join the production herd. A cow typically remains in the herd until about five years of age, although many cows are capable of remaining productive in the herd for 12 to 15 years. Each period of production or lactation lasts for 11 to 14 months or longer and spans the time period from calving to dry-off, which is when milking is terminated about 40 to 60 days before the next anticipated calving. Thus, cows are bred while they are producing milk, usually beginning at about 60 days after calving, to maintain a yearly calving schedule. Following the 35 to 60-day dry period, the cow calves again, and the lactation cycle begins anew. Cows average about 2.8 lactations, although many remain productive considerably longer (Hare et al., 2006).

  Lactating cows may be housed in tie stall (stanchion) barns, which limit the cows' mobility because the cows are tethered, fed, and milked in the stalls. A gutter is used to remove the manure by a barn cleaner, which typically places the manure directly into a manure spreader or in a temporary storage pile. Freestall barns allow the cows to move freely in and out of stalls, and the cows are moved to a separate area (milking center or parlor) for milking. Manure typically accumulates in alleyways and is removed via scraping, vacuuming, or flushing with either clean or recirculated water. Some freestall barns have slotted floors with long-term manure storage below the floors. Manure is generally worked naturally through the slots by the cows' feet and with assistance via mechanical scraping equipment. Dairy facilities may also use pastures and dry-lots to house lactating cows. Lots are scraped periodically, as are pastures occasionally, and the solid manure is collected. Although not prevalent, some dairy facilities may house lactating cows in bedded pack or compost barns, again handling manure as a solid material.

- **Dry Cows.** Much like growing cows, housing options for dry cows are the same as described above for lactating cows. The key determinant is management preference for the farm owner and/or facility availability.

Manure and soiled bedding from barns can be handled in a number of ways. Manure can be removed from the barns mechanically and directly loaded into manure spreaders, although this is not common on medium and large farms. Manure can also be processed in an anaerobic digester where bacteria can break down manure to produce biogas that can be flared or captured for energy purposes prior to storage of digester effluent. When manure has a lower solids content, it may be stored in a tank or pit as a slurry, or transported to a solid-liquid separation system with the liquid fraction conveyed (pumped or by gravity) to a long-term storage pond, while the solids can be dewatered naturally and reused as bedding, composted, land-applied, and/or sold. In dry-lot systems, the manure in the pens is typically stacked and following storage is either land-applied or composted. Lot runoff and milking parlor wash water is pumped to a storage pond. There are some dry-lot dairies that use a flush system to clean manure from alleyways behind the feed bunks; this washwater is eventually stored in a wastewater pond. Open freestall dairies have a combination of barns with exercise yards between the barns, and therefore manure is handled similarly as in a traditional freestall barn and dry-lot production system. Wastewater from milking centers (manure, clean-in-place water, and floor washdown water) is typically combined with barn manure destined for long-term storage, and may go through a solid-liquid separation process first. In pasture-based systems, manure is deposited directly onto the pasture and therefore not intensively managed, but may accumulate in areas where animals tend to congregate (e.g., watering areas, shade).

5.2.2 Beef Production Systems

5.2.2.1 Overview of Beef Production Systems

The U.S. beef production system is comprised of several key components for beef cattle, their waste, and their end products, as depicted in Figure 5-4. This conceptual model provides an overview of the typical beef processing systems, following the segments of the beef cattle industry (i.e., cow-calf, stocker, feeder/finisher, and packer) from birth to slaughter and following waste from the animal through a management system. Waste is produced during each stage of activity occurring in the system, and depending on the location, is managed differently.

Of the 90 million beef cattle in the United States, approximately 50 million are mature cows and their calves on cow-calf operations (USDA NASS, 2012), which range in size from a few cows to several thousand cows. These operations are normally based on forages, either improved pastures or native range, and vary in size from a few acres to hundreds of sections. Typically, when calves are 150 to 220 days of age they are weaned and moved to pasture for periods of 60 to 200 days (the stocker phase), although some may move directly to a feedlot. The pastures may be native range, improved perennial pastures, or annuals such as wheat pasture, forage-sorghums, and crop residues such as corn stalks. After the stocker phase, calves normally move to feedlots where they are fed grain- and byproduct-based diets for 110 to 160 days, until they are ready for harvest. In addition, steers and cull heifers from dairy operations are also fed. Approximately 23 million cattle are fed in feedlots annually in the United States. Feedlots range in size from a few hundred head to more than 100,000 head capacity.

5.2.2.2 Diet Information for Beef Cattle

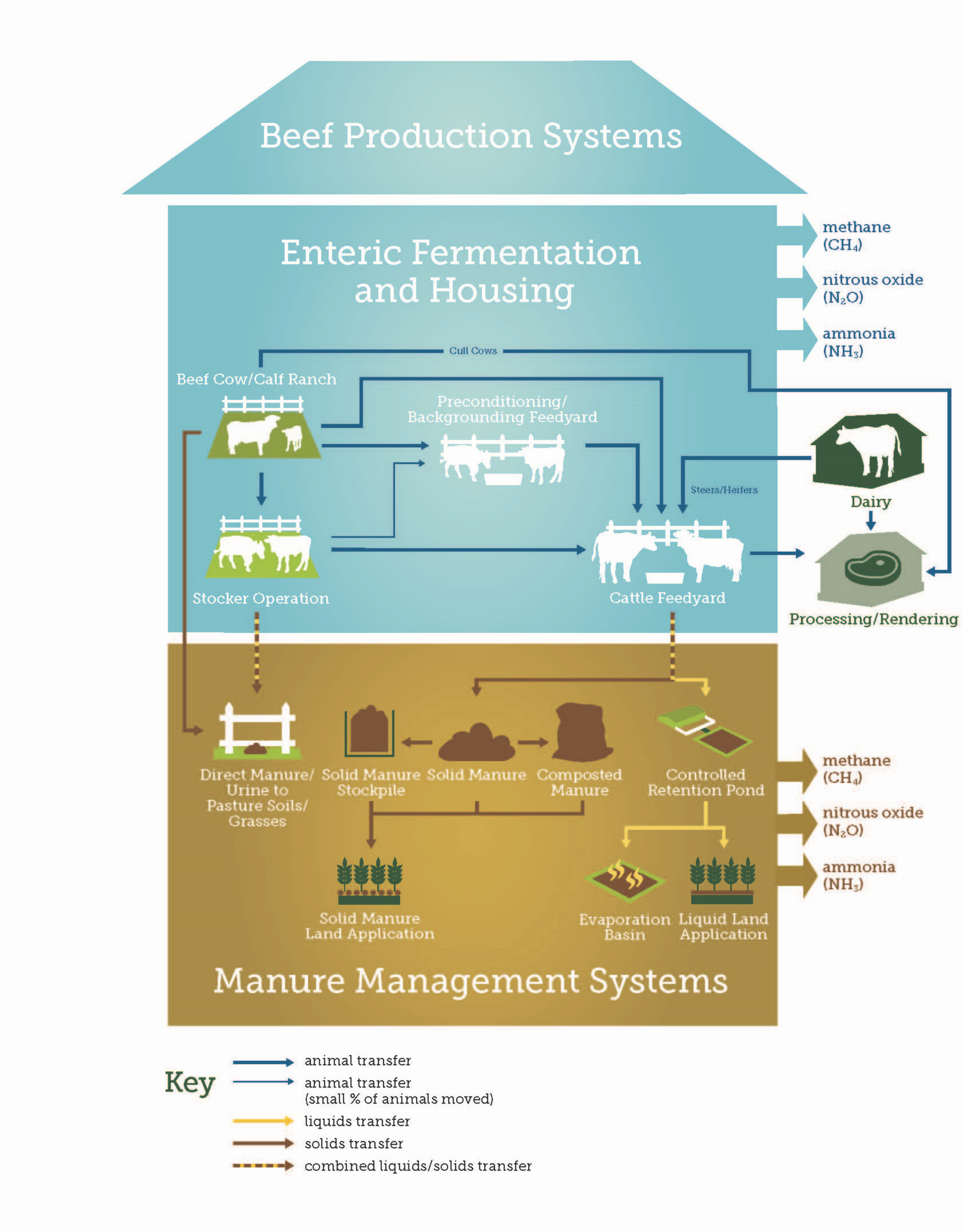
Cow-Calf and Bulls

Grazing pastures may be native range, improved perennial pastures, or annuals such as wheat pasture, forage-sorghums, and crop residues such as corn stalks. Beef cows and bulls are typically fed supplemental feeds during times when pasture or range forage does not meet their nutritional requirements, usually in winter. A recent survey of the beef cow-calf industry found that 74 percent of operations fed a protein supplement and 51 percent fed an energy supplement (USDA, 2010). Overall protein was supplemented for an average of 173 days (SE=9.6) and energy for 162 days (SE=12.7), but this was highly variable across regions of the country. Ninety-seven percent of operations in the survey supplemented the cow herd with roughage for an average of 154 days (SE=7.0). The protein supplements were reported as plant protein or urea-based. Corn was reported as the primary energy supplement. The amount of supplement fed per head per day was not included in the report.

Stockers

Stockers graze forage, including wheat pasture, improved pastures, range, and crop residues. Stocker cattle may also receive supplemental protein or energy feeds to increase performance and/or extend pasture forage. Supplements may or may not contain an ionophore. Some stocker calves may be implanted with a growth promoting implant; others are not.

Figure 5-4 Conceptual Model of Beef Production Systems in the United States



Feedlot

Cattle typically enter feedyards between the ages of 100 and 350 days weighing 200 to 350 kg, and go to slaughter weighing between 500 to 700 kg. They are fed high-concentrate or high-byproduct diets for 100 to 200 days. Of the cattle fed, approximately 55 percent are beef steers, 25 to 30 percent are beef heifers, and 12 to 20 percent are dairy steers and heifers. The vast majority of cattle fed are beef breeds of British or Continental breeding. However, many cattle with Brahman genetics are also fed, mostly in the southern plains. In areas with a significant dairy industry, steers and heifers of dairy breeding (mostly Holstein) are also fed.

Typical feedlot diets contain high concentrations of grain (75 percent or more) and/or byproducts such as distillers grains and gluten feed. They are normally balanced for protein, energy, vitamins, and minerals (Vasconcelos and Galyean, 2007). Because many byproducts contain high concentrations of protein and minerals such as phosphorus and sulfur, when these byproducts are fed, dietary concentrations of protein and some minerals may exceed animal requirements. Feeding of ionophores such as monensin is common in the United States, as is the use of growth-promoting implants. The diets fed in feedyards tend to differ between the northern and southern plains. Finishing diets based on dry-rolled corn (DRC) and high-moisture corn (HMC) dominate in the North, whereas diets based on steam-flaked corn (SFC) dominate in the South. The use of bioethanol co-products such as distiller's grains and corn-milling co-products such as corn gluten feed in finishing diets is greater in the northern plains because of the greater availability of these co-products, but their use is increasing in the southern plains.

5.2.2.3 Beef Cattle Housing and Manure Handling

Cow-Calf and Bulls

Cow herds and replacement heifers are most often housed on pasture. Feces and urine are deposited on pastures and rangeland and may be concentrated in areas in which feeding or watering takes place.

Stockers

Stockers are usually housed on pasture and thus no manure handling is used and GHG emissions are a part of the croplands section (see Chapter 3, Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems). Calves to be used as stockers can be housed for short periods of time in dry-lots.

Feedlot

Housing and manure management at most beef cattle feeding operations differ greatly from those used in other livestock species, with the vast majority being finished in dry-lot pens with soil surfaces. Manure is normally deposited on the pen surface and scraped from the pens after each group of cattle goes to market. Part of the manure may be stacked in the pen to provide mounds that improve pen drainage and assure that cattle have a dry place to lie after rains. Manure removed from the pen may be immediately applied to fields near the feedlot, stockpiled for later use, or composted in windrows. Manure scraped from the pens normally has a moisture content of 30 to 50 percent and may contain some soil from the pen. Because the manure may remain in the pen or in stockpiles for several months before it is applied to the field, a portion of the nitrogen and carbon may be lost before the manure is collected or applied to land. Runoff from pens is normally collected in retention ponds. Settling basins may be used to limit the quantity of manure solids and soil particles that reach the retention pond.

In the Northern United States, and in areas with high rainfall, cattle may be fed in naturally ventilated barns with slotted floors for collection of urine and feces or in deep-bedded barns with concrete floors in which the manure and bedding (normally straw or stalks) are allowed to accumulate during the feeding period (Spiehs et al., 2011). Adding bedding will increase the quantity of carbon (and possibly nitrogen) available to be metabolized by microbes in the pen. These facilities are characterized by the absence of runoff control systems.

5.2.3 Sheep Production Systems

5.2.3.1 Overview of Sheep Production Systems

There are 81,000 sheep and lamb operations in the United States, with an inventory of 5.53 million sheep and lambs as of January 1, 2011 (USDA NASS, 2011). Most breeding flocks are small and consist of less than 100 head of ewes. The lamb feeding industry is also diverse in size, with small feedlots located throughout the farm flock areas and large feeding operations located in close proximity to local grain production capacity (Shiflett, 2011).

5.2.3.2 Diets, Housing, and Manure Handling for Sheep

Lambing season may occur at various times during the year, depending on production objectives, feed resources, environmental conditions, and market targets. When lambing occurs, January through March, ewes are generally housed in bedded barns. Bedding is removed and spread after animals are turned out on pasture. Ewes are generally bred on pasture in September through November and, depending on weather, will be moved into barns prior to lambing—or earlier as forage availability and weather dictate. Diets consist of pasture or grazing crop residue from spring turnout through early- and mid-gestation. When grazed forage is no longer available, ewes are housed or moved to dry-lots and fed hay and/or hay and grain diets as gestation requirements dictate. The primary forage source is alfalfa, and corn is the predominant grain. Diets range from 100 percent hay to 60:40 percent forage:concentrate while lactating. Most lambs are weaned at approximately 90 days and 41 kg and sent to feedlots for finishing.

Pasture lambing is another farm flock production system that is used to maximize nutrients provided by grazed forages. In this case the ewe is bred in November or December to lamb on pasture in April or May. Lambs are weaned at approximately 120 days and 32 kg and may be sent to the feedlot or finished on grass. Ewes are not fed grain, and harvested forage is provided only when growing seasons and weather dictate. These flocks will be housed in bedded barns in areas requiring protection from winter weather conditions. Range production systems include lambing in April or May, where most (and in some cases all) diets are provided by grazed forages. Supplementation with harvested feeds or grains is usually in response to unpredictable weather and environmental conditions.

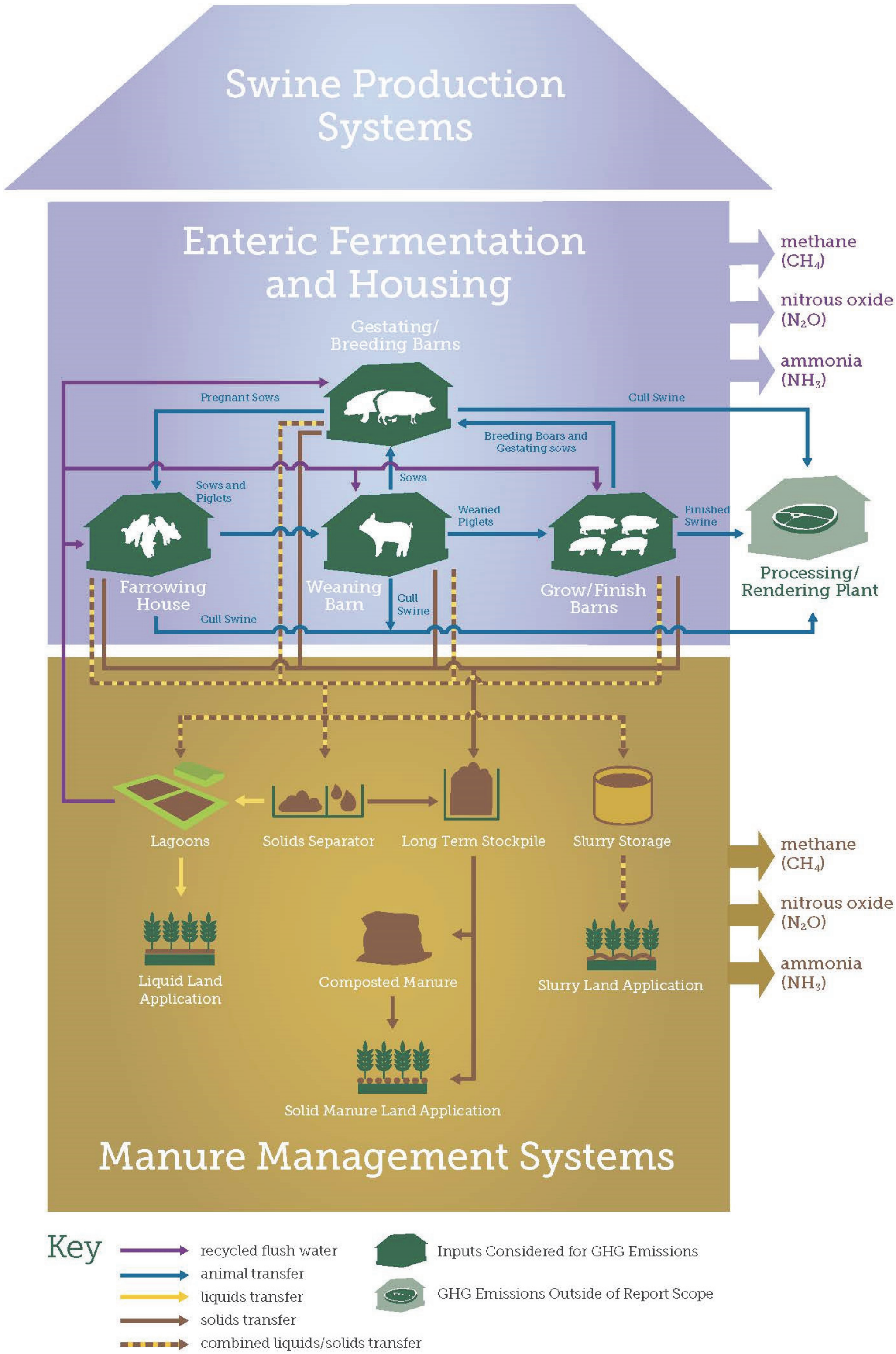
Most lambs are finished in feedlots and fed diets containing 85 to 90 percent grain. Length of feeding periods will range from weeks to months depending on in-weights and time required to reach final weight (industry average final weight = 61 kg). Sheep feedlots are primarily dry-lots, and manure is scraped from the pens similarly to beef cattle feedlots.

5.2.4 Swine Production Systems

5.2.4.1 Overview of Swine Production Systems

The conceptual model (Figure 5-5) of the U.S. swine production system provides an overview of typical production systems, following animals from birth to harvest and following manure from the animal through a management system. Manure is produced during each stage of production occurring in the system, and depending on the location, is managed differently. This has implications on the quantity of GHG emissions and sinks, some of which are discussed in detail in the emissions discussion section (Section 5.3.4).

Figure 5-5: Conceptual Model of Swine Production Systems in the United States



Swine production in the United States remains important to both the nation's diet and economy (Davies, 2011), with significant levels of consumption, imports, and exports. According to the U.S. Department of Agriculture's National Agricultural Statistics Service, the 2011 population was nearly 66 million head (USDA NASS, 2012).

Swine are predominantly grown with production of pork occurring in a two-stage or three-stage system:

- Stage 1: Sow operation, piglets leave at weaning.
- Stage 2 (optional): Nursery operation, weaning (10 days of age/17 lbs) to 42 days of age/45 lbs.
- Stage 3: Several options:
  - A finishing operation (16-week production site where piglets are delivered from a nursery site at approximately 42 days of age/ 45 lbs and stay until 154 days of age (22 weeks) or
  - A wean-to-finish operation (24-week production site where pigs are delivered at weaning directly from a sow operation (10 days of age/17 lbs) and stay until 178 days of age (25.5 weeks)).

The manure management systems associated with these production operations all have the basic elements of collection, storage, treatment, transport, and utilization. Most swine facilities handle manure as a slurry either within the building (deep pit finishing barns or shallow pit nursery, gestation or finishing barns) or in outside storage (pull-plug systems for nurseries, sows, or finishing pigs). Collection and storage is generally accomplished by storage of the waste under the facility, discharge to a separate storage tank, or flushing to an anaerobic lagoon. In the case of in-house manure storage, little water is added to the storage structure, and anaerobic conditions prevail with little biological processing of manure taking place. Outside storage structures that contain slurry with little dilution water offer minimal biological treatment as well. However, lagoon systems where manure is flushed from housing and additional dilution water is added offer more treatment. Dry systems or deep-bedded systems exist to a much lesser extent, primarily for sow or finishing production. In these cases bedding material, often straw, is provided and manure plus bedding is handled as solid material, sometimes composted.

In the Midwest, the system of moving stored swine waste to crop fields is well defined and understood (Hatfield and Pfeiffer, 2005; Malone et al., 2007; Jarecki et al., 2008; Vanotti et al., 2008; Brooks and McLaughlin, 2009; Jarecki et al., 2009; Agnew et al., 2010; Cambardella et al., 2010; Lovanh et al., 2010). Yet these systems continue to evolve to address both old and new issues, such as frozen ground, application timing, and emissions associated with soil application via new equipment. All of the manure management systems result in GHG emissions, but they vary in terms of point and non-point sources.

5.2.4.2 Diet Information for Swine

The swine industry feeds primarily a corn-soybean meal based diet. Dried distillers grains with solubles (DDGS) are often fed to both sows and finishing pigs and, as availability of this feed increases, the amount fed increases to as much as 40 percent of diet dry matter intake (DMI). Similarly, when synthetic amino acid sources price competitively with feed protein sources, the number of synthetic amino acids included in finishing pig diets increases. Two (lysine and methionine) or more (threonine, perhaps tryptophan) synthetic amino acids are fed commonly today with the benefit of reducing total nitrogen fed, and therefore excreted, by swine.

5.2.4.3 Swine Housing and Manure Handling

Most commercially-raised finishing swine are housed indoors to provide a biosecure environment and reduce disease pressures. Manure is handled as slurry with little or no bedding added to the system and minimal addition of water. A small but growing portion of the commercial swine industry house both finishing pigs and sows in hoop barns. In these cases, bedding material, often straw, is provided, and manure plus bedding is handled as solid material.

5.2.5 Poultry Production Systems

5.2.5.1 Overview of Poultry Production Systems

The U.S. poultry production system is comprised of several key processes for poultry, their manure/litter, and their end products (meat, eggs) as depicted in Figure 5-6.

The figure provides an overview of the typical production systems, following both the layer and broiler phases. This conceptual model provides an overview of the typical poultry production systems, following birds from birth to slaughter and following manure from the animal through a management system. Manure is produced during each stage of activities occurring in the system, and depending on the location, is managed differently. The emissions from manure management are discussed in detail in Section 5.3.

The U.S. poultry industry is the world's largest producer and second largest exporter of poultry meat. The U.S. is also a major egg producer. The poultry and egg industry is a major feed grain user, accounting for approximately 45.4 billion kg (100 billion lbs) of feed yearly.

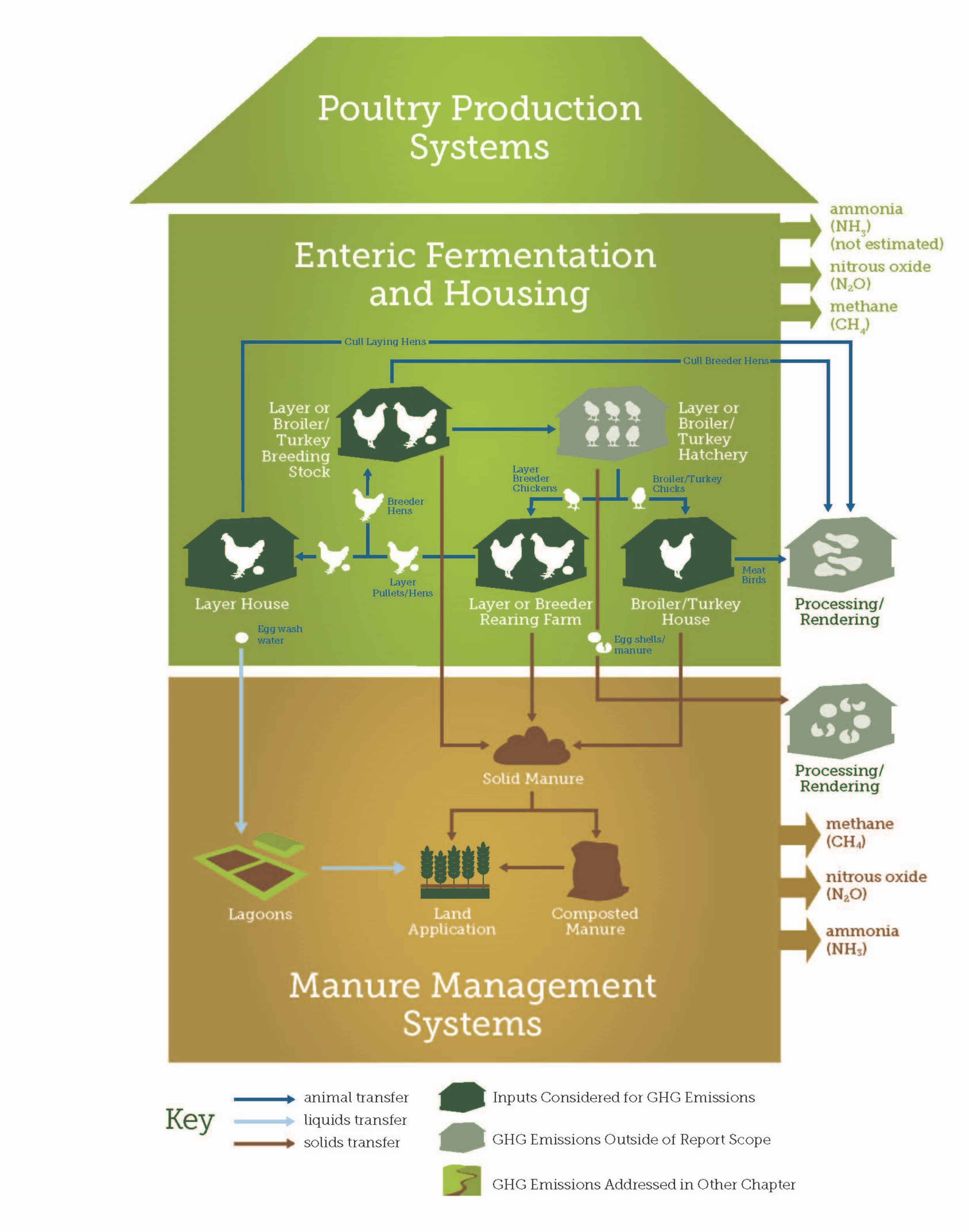
The egg incubation period for a chicken is 21 days. Following hatch, broiler chickens are reared for 42 to 49 days (six to seven flocks per year), depending upon the market intent (e.g., roasters). U.S. egg operations produce more than 90 billion eggs annually. More than 75 percent of egg production is for human consumption (the table-egg market). The remainder of production is for the hatching market. These eggs are hatched to provide replacement birds for the egg-laying flocks and to produce broiler chicks for grow-out operations. Following a 16 to 22 week growth period, hens start laying eggs.

The U.S. turkey industry produces more than one-quarter of a billion birds annually, with the live weight of each bird averaging more than 25 lbs. The egg incubation period for a turkey is 28 days. Following hatch, turkey poults are reared for 15 to 22 weeks (one to three flocks per year) depending on the market intent (e.g., roasters).

5.2.5.2 Diet and Growth Information for Poultry

Diets for meat birds consist largely of corn and soybean meal (commonly 85 to 92 percent of the diet); however, alternate ingredients such as dried distillers grains with solubles (DDGS) and other co-products, and synthetic amino acids are increasingly used. Hen diets are most commonly composed of corn and soybean meal. Other ingredients, such as DDGS, may be included (rarely more than 20 percent of the diet). Ingredient variability is largely in sources of supplemental energy, minerals, and additives to improve animal health and performance. Diets are formulated based on growth rate and egg production and fed as either a mash or a pellet. Bone strength is an important characteristic of meat bird quality therefore provision of minerals such as calcium and phosphorus are carefully considered when diets are formulated. Similarly, eggshell quality is key for laying hens, and as a result, calcium utilization is a key element in diet formulation.

Figure 5-6: Conceptual Model of Poultry Production Systems in the United States



Poultry breeds change rapidly, demonstrating improved production efficiency, and as such, diets are increasingly dense with energy and protein. These changes are due to a combination of genetics and management, including diet formulation.[2] While diet and genetic influences were considered in a study by Havenstein et al. (2007), the results suggest that the diet changes that occurred between 1966 and 2003 interacted with other factors (flock age, ambient temperature) to influence bird growth. Some estimate that 85 percent of the improvement in the growth rate of broiler chickens is attributable to genetics (Havenstein et al., 2003).[3]

In the United States there is no ban, at present, on use of antibiotic growth promoters (AGPs) in poultry production (meat birds). However, the trend is toward consumers wanting products that have not used AGP. Finding replacements for AGP will likely involve the use of multiple products in the diet, each with some of the benefits of AGP, and management changes will play a key role in maintaining animal productivity in their absence. It is unlikely that a single replacement will be found that will prove to be economically viable (Dibner and Richards, 2005).

5.2.5.3 Poultry Housing and Manure Handling

The vast majority of the industry raises birds on litter in mechanically ventilated or naturally ventilated houses. Reuse of litter and number of flocks grown on the same litter is variable across the country, and can range from as low as a single flock to as many as 18 flocks on the same litter source. Litter dry matter content can vary from 40 to 80 percent, depending on management.

Laying hen and pullet housing types range from high-rise houses where hens are in cages and manure accumulates in a basement under the cages and is removed annually, to a manure-belt house where hens are in cages and manure is removed daily or more frequently from the basement to an external shed and stacked before periodic removal for land application (once or twice per year), to aviaries where hens are raised on litter (in large rooms as opposed to cages) that is removed from the aviary annually or more frequently. When manure is removed from the house it may be immediately applied to fields, stockpiled, or composted. Moisture content may vary from 80 percent moisture down to 20 percent moisture (aviaries).

5.3 Emissions from Enteric Fermentation and Housing

Emissions from animal production systems include those from both enteric fermentation and from animal housing (including animal manure in housing areas that may ultimately be flushed or scraped and then transported to an external manure management system). The production of GHGs in livestock systems originates from a variety of sources, including directly from the animals themselves; manure in lots and barns; stockpiled and composting manures; manure slurries or waters in tanks, pits, lagoons, retention ponds, settling cells, etc.; and from soils after manure application. Emissions from these sources depend on animal size and age, diet, manure production, handling and storage system, lot surface and soil characteristics, and ambient weather conditions (i.e., temperature, wind, humidity, and precipitation). For each animal type, this section summarizes

[2] Havenstein et al. (2007) compared 1966 strains to 2003 strains and observed a 20 percent better cumulative feed conversion ratio in the 2003 tom turkey fed a 2003 diet relative to a 1966 tom fed a diet typical of 1966. Feed efficiency to 11 kg bodyweight was approximately 50 percent better (2.13 at 98 days of age in 2003 toms, compared with 4.21 at 196 days for 1966 toms).

[3] Havenstein et al. (2003) compared the 1957 Athens-Canadian Randombred Control strain and the 2001 Ross 308 strain of broilers when fed representative 1957 and 2001 diets. The 42-day feed conversions for the Ross 308 birds fed the 2001 and 1957 feeds were 1.62 and 1.92, respectively (with average bodyweight of 2,672 and 2,126 g). The 42-day feed conversions for the Athens-Canadian Randombred Control were 2.14 and 2.34 (average bodyweight of 578 and 539 g, respectively).

the current understanding of enteric fermentation and livestock housing emissions and presents recommended models for estimating such emissions, including the rationale for selecting methods.

Actual field measurements of GHGs from enteric fermentation over the past several decades have been instrumental in improving our understanding of the underlying science and the resulting models presented in this section. For dairy animals, most of the emissions estimates available represent the lactating animal. The equations for growing beef animals are likely appropriate for growing dairy animals if diet composition is considered. The text boxes on the following pages summarize several of the key techniques that have been used in measurement studies for both individual animals and groups of animals. Further studies of this type will be needed to address research gaps in Section 5.5.

This section provides the recommended method for estimating GHGs from enteric fermentation and applicable housing emissions. Quantitative methods are provided for dairy, beef, sheep, swine, poultry, and other animals (i.e., goats, American bison, llama, alpacas, and managed wildlife). For each section, background information is provided on the range of emissions and existing models for estimating emissions and the rationale for the method selected. For estimating emissions from enteric fermentation, the activity data is the same for all animal types. Ancillary data includes the properties of the diets (e.g., crude protein (CP), digestible energy (DE), neutral detergent fiber (NDF)). For simplicity, activity data and ancillary data are listed in Table 5-2 and are not repeated below for each animal type.

5.3.1 Enteric Fermentation and Housing Emissions from Dairy Production Systems

Although the dairy industry is primarily composed of three livestock types [growing (i.e., calves, replacement heifers), lactating cows, and dry cows], most of the limited emissions research conducted to date has been targeted at lactating cows, which typically produce at least 50 percent more enteric CH_4 per head than other dairy cattle. Few emissions data exist for calves, heifers, and dry cows. Therefore, the discussion here focuses primarily on lactating cows.

Data needed to estimate emissions include housing system (pasture, barn type, dry-lot), animal characteristics (breed, body weight, growth potential, stage of lactation, milking frequency, and milk production) and population, dietary information (DMI, dietary CP—also NDF, fat, DE, metabolizable energy (ME), net energy (NE), nutrient excretion (N, C, and volatile solids), use of recombinant bovine somatotropin, use of monensin, type of manure handling system, frequency of manure removal, type of bedding, and manure characteristics (total ammonium nitrogen, pH).

Enteric Fermentation

Enteric CH_4 production varies with production stage in dairy cattle, with the highest rates being produced by lactating cows (Table 5-4). This table illustrates, conceptually, the observed variation in cattle at different stages of maturity and activity, but it is not intended to provide a depiction of absolute differences. There are many factors that affect enteric CH_4 production, and therefore altering dairy cattle diets could have an impact on enteric CH_4 production. For an in-depth discussion of dietary effects on enteric CH_4 production, see Section 5.3.7 (*Factors Affecting Enteric Fermentation Emissions*). However, the results in Table 5-4 clearly illustrate the difference in enteric emissions; in particular, emissions from dairy cattle are relatively higher than those from growing (i.e., heifers) and dry cattle.

Table 5-4: Examples of CH_4 Emissions Measured in Dairy Cattle

Animal Type	CH_4 Emission	Method Used to Measure Emissions	Reference
Dairy cattle	260 g animal^{-1} day^{-1}	Calculated Blaxter and Clapperton	Crutzen et al. (1986)

Animal Type	CH_4 Emission	Method Used to Measure Emissions	Reference
Heifer 6-24 month	140 g LU^{-1} day^{-1}	See above	
Dairy cattle, dry period	139 g LU^{-1} day^{-1}	Respiration calorimetry	Holter & Young (1992)
Dairy cattle, lactating	268 g LU^{-1} day^{-1}	See above	
Dairy cattle	257 g LU^{-1} day^{-1}	Respiration calorimetry	Kirchgessner et al. (1991)
Dairy cattle, lactating	429 g $animal^{-1}$ day^{-1}	Wind tunnel	Sun et al. (2008)
Dairy cattle, dry period	290 g $animal^{-1}$ day^{-1}	Wind tunnel	
Dairy cattle, lactating	538 – 648 g $animal^{-1}$ day^{-1}	Respiration calorimetry	Aguerre et al. (2011)

LU, livestock unit = 500 kg

Methods for Measuring CH_4 Emissions from Enteric Fermentation

Individual Animals

The standard method of measuring CH_4 emissions from ruminants is by respiration calorimetry chambers. Other techniques, including head boxes, internal tracers, micrometeorology, isotope dilution, and polyethylene tunnels, have been used (Kebreab et al., 2006; Harper et al., 2011). Several new technologies have been developed to measure individual animal emissions. To address the difficulty in measuring enteric CH_4 from many animals on pasture, alternate methods are sought. As one example, Goopy et al. (2011) has proposed a portable static chamber method to measure daily CH_4 production. Until validated, results using alternate methods should be viewed with caution.

A variety of respiration chambers have been developed to measure enteric CH_4 losses and/or total energy metabolism of the animal. In general, air is pulled from the chamber at a known rate and replaced with outside air. Flow of air and concentrations of CH_4, CO_2, and oxygen (O_2) in the air entering and leaving the chamber are measured to determine total CO_2 and CH_4 production and O_2 consumption. When properly calibrated and used, respiration chambers give highly accurate, precise measurements. However, they are expensive to build and operate, and require significant knowledge, skill, and labor.

Feed intake and production are usually depressed in animals in chambers and the measurements do not necessarily reflect intake and production from typical commercial operations. This limitation can be partially overcome by feeding animals at different levels of intake and measuring the effects of intake level. Head boxes use the same principles as respiration calorimetry, and have many of the same limitations. In-barn chambers using drop-down curtains have been used to measure, at relatively low cost, emissions of NH_3, CH_4, and other gasses from groups of dairy cows (Powell et al., 2007; Powell et al., 2008; Aguerre et al., 2011).

Internal tracer techniques such as the sulfur hexafluoride (SF_6) tracer method (Johnson et al., 1994) were developed to allow measurements from free-ranging animals, such as those managed under pasture situations, or when real-world levels of feed intake are needed. The limitations to this method are the need for trained animals, the need for larger sample sizes (compared with chambers) to detect the influence of mitigation techniques, and concerns about inconsistent releases of tracer gas from SF_6 permeation tubes manufactured for large release rates. Additionally, the SF_6 technique generally results in emission estimates that are lower than chamber measurements; possibly because the SF_6 method does not measure all lower gut CH_4 production (McGinn et al., 2006). The advantages and shortcomings of the SF_6 method have been recently reviewed (Lassey et al., 2011).

Methods for Measuring CH_4 Emissions from Enteric Fermentation

Group of Animals

Micrometeorology methods have been used extensively to measure CH_4 and NH_3 emissions from pastures, whole feed yards, or portions of the feed yard (pens, retention ponds, manure stockpiles, etc.). These methods have been reviewed (Fowler et al., 2001; Flesch et al., 2005; Harper et al., 2011). Lauback et al. (2008) compared the SF_6 method with three micrometeorological methods (integrated horizontal flux, flux gradient, and backward Lagrangian stochastic (bLS)) using steers grazing paddocks. In general, the micrometeorological methods gave higher CH_4 measurements than the SF_6 method, with the difference being greater when animals were within 22 meters of the CH_4 sampler. This effect was especially true for the flux gradient method. The lower values for the SF_6 method could be due in part to the fact that the SF_6 method does not measure emissions from the lower gut or from fermentation of feces on the paddock surface.

Tomkins et al. (2011) compared enteric CH_4 emissions of steers on pasture using the bLS method and respiration chambers. Emissions estimated using the bLS model were slightly greater than with respiration chambers (136.1 vs. 114.3 g head daily^{-1}). However emissions per gram of DMI were similar (29.7 vs. 30.1 g CH_4 kg DMI^{-1}, respectively), suggesting that the bLS model may be suitable for estimating enteric emissions.

Most dispersion models and micrometeorological methods assume that emissions are uniformly distributed over the source area. In some cases, such as for individual cattle in a pen or field, this is not true. Therefore, McGinn et al. (2011) developed a method that used a point-source dispersion model and atmospheric CH_4 concentrations measured using multiple open-path lasers to measure CH_4 emissions from a paddock containing 18 cattle. Measured enteric CH_4 emissions were similar to values measured using other techniques. However, recoveries of known CH_4 releases averaged only 77 percent using this method. The method gave more reliable measurements during the daytime when atmospheric conditions were unstable than at night when atmospheric conditions were stable.

Methods for Measuring Emissions from Manure

Estimating emissions from large open source areas typically associated with both dairy and beef cattle production is very challenging, due to the inability to contain and measure the source area. Instruments and techniques to measure ambient atmospheric gases from these large source areas (i.e., dry-lot beef and dairy cattle yards, freestall dairies with naturally ventilated curtain sidewall barns, and grazing land) must be able to detect lower concentrations than those encountered in typical enclosed confined animal production systems, because of the low concentrations and high variability resulting from high and variable ventilation rates. A larger challenge with measuring emissions from open facilities is the ability to estimate airflow due to the lack of a defined, constant air inlet and air outlet. Reported background NH_3 concentrations typically range from <1.3 to 53.3 parts per billion (ppb) (Todd et al., 2005), background atmospheric N_2O concentrations near feedyards average about 319 ppb (Michal et al., 2010), and background CH_4 concentrations typically run in the area of 1,780 ppb (Michal et al., 2010).

Methods for Measuring Emissions from Manure (Continued)

Numerous factors can affect atmospheric concentrations of NH_3 and GHG near livestock operations including sampling height, atmospheric stability, wind speed, background concentrations, stocking density, sampling site, sampling time, temperature, and wind direction (fetch). Average daily NH_3 concentrations measured at a variety of similar source areas ranged from approximately 100 to 2,000 µg m^{-3}. Measured maximum concentrations rarely exceed 2,000 µg m^{-3}. Ammonia concentrations decrease rapidly downwind of source areas (Miner, 1975), approaching background concentrations in less than 800 meters (McGinn et al., 2003; Sweeten, 2004).

Atmospheric CH_4 concentrations measured at feedlots and dry lot dairies have ranged from 3.3 to 4.7 parts per million (ppm) (Michal et al., 2010), and from background (approximately 1.78 ppm) to 6.20 ppm (Bjorneberg et al., 2009), respectively. Nitrous oxide concentrations measured at feedlots ranged from 319 ppb (background) to 443 ppb and averaged 396 ± 16 ppb (Michal et al., 2010). Nitrous oxide concentrations were highest following a rainfall event. After a rain, CH_4 concentrations averaged 3.7 ± 0.1 ppm. At dry-lot dairies, median N_2O concentrations ranged from 314 ppb to 330 ppb, which are very close to global background values (Bjorneberg et al., 2009).

Small flux chambers and wind tunnels have been used to estimate emissions of NH_3, CH_4, and N_2O from farmlands, pastures, pen surfaces, lagoons, and retention ponds (Hutchinson and Mosier, 1981; Venterea et al., 2009; Venterea, 2010; Harper et al., 2011; Hristov et al., 2011). In general, chambers alter the microenvironment of the surface and may alter emissions. Thus, the accuracy of these methods for determining emission factors for some gases (especially NH_3) has been questioned (Gao and Yates, 1998; Harper, 2005; Venterea et al., 2009; Parker et al., 2010; Venterea, 2010; Harper et al., 2011). Measures of NH_3 emissions using flux chambers and wind tunnels are highly dependent upon air flow and air turnover rates in the chamber (Cole et al., 2007b; Parker et al., 2010). Based on the conventional two-film model used to describe volatilization from a solute-solvent mixture (Parker et al., 2010), many gaseous emissions are controlled by the gas film above the liquid or the upper portion of the liquid (liquid film) defined by the Henry's law constant. If volatilization is inhibited by high concentrations in the gas phase (i.e., gas-film controlled), increases in gaseous concentration—such as with flux chambers—will lead to significant underestimation of true flux. Venterea (2010) reported that emissions of N_2O estimated using static chambers were underestimated by approximately three to 38 percent, depending upon soil water content, type of regression performed (linear vs. quadratic vs. nonlinear), and other factors. The percentage of underestimation tended to be greater with dry soils, probably because N_2O flux is lower when soils are dry. Sommer et al. (2004) reported that GHG emissions from compost stockpiles measured using static chambers were only 12 to 22 percent of values measured using the integrated horizontal flux method.

Because of these factors, flux chambers should be used to examine relative differences, rather than emission factors of NH_3, CH_4, and N_2O emissions from pen surfaces, lagoons, retention ponds, manure stockpiles, or compost windrows. In addition, the surface of pastures and feedlot pens is temporally and spatially heterogeneous, with dry areas, areas with fresh feces, and areas with urine of different ages (Woodbury et al., 2001; Cole et al., 2009a; Cole et al., 2009b). To adequately represent the surface, the number of chamber measurements required (estimated as the coefficient of variation squared/100: Kienbusch, 1986) can be very large (i.e., one chamber/quare meter: Cole et al., 2007b).

Housing

There are a wide variety of dairy cattle housing systems due to variations in herd size and regional practices. In the northeastern United States, herd size tends to be smaller and cattle are housed in freestall and tie-stall barns and on pasture; in the western part of the country, herd sizes tend to be larger and animals are housed in freestall barns or dry-lots with few producers using pasture-based systems. These differences in housing can lead to differences in both GHG and NH_3 emissions. Examples of reported emissions from varying housing systems are presented in Table 5-5.

Table 5-5: Examples of Reported On-Farm Emission Estimates for CH_4, N_2O, and NH_3 from a Variety of Dairy Cattle Housing Systems

Housing	Country	Emissions (g cow^{-1} d^{-1})			Reference
		CH_4	N_2O	*NH_3*	
Barn	Germany	402		*64.8*	Saha et al. (2014)
Tie stall barn	Austria	170-232[a]	0.14-1.2[a]	*4-7.4*[a]	Amon et al. (2001)
Barn	Germany	256	1.8	*14.4*	Jungbluth et al. (2001)
Dry-lot	U.S.			*41-140*	Cassel et al. (2005)
Hardstanding	UK	0.03[b]	0.01	*11*	Ellis et al. (2001)
Open-freestall	U.S.	410	22	*80*	Leytem et al. (2013)
Tie stall barn	Canada	390			Kinsman et al. (1995)
Pasture	NZ	300-427			Laubach & Kelliher (2005)
Dry-lot	U.S.	490	10	*130*	Leytem et al. (2011)
Standoff pad	NZ	1.66[b]	0.03		Luo & Saggar (2008)
Barn	Denmark	256	1.2	*16*	Zhang et al. (Zhang et al., 2005)
Dry-lot	China	397	37		Zhu et al. (Zhu et al., 2014)
Barn	Sweden	216-312[a]		*21-27*[a]	Ngwabie et al. (2009)
Barn	Germany	464	45	*92.4*	Samer et al. (Samer et al., 2011)
Pasture	Uruguay	372			Dini et al. (Dini et al., 2012)

*Denotes measurements in g LU^{-1} d^{-1}, where a LU (livestock unit) = 500 kg.
†Measurements do not include enteric CH_4 production.

Variations in emissions from housing are due to factors such as temperature, diet composition, water consumption, ventilation flow rates, type of manure handling systems, manure removal frequency, feces, and urine characteristics (i.e., pH and total ammoniacal nitrogen (TAN)), and type of bedding used. Although differences can be great between emission rates, there are some emission characteristics that are consistent across most studies. Many studies have reported strong diel trends in emissions of CH_4 and NH_3, with emissions tending to be lower in the late evening and early morning and then higher throughout the day till early evening (Amon et al., 2001; Cassel et al., 2005; Powell et al., 2008; Sun et al., 2008; Bjorneberg et al., 2009; Flesch et al., 2009; Ngwabie et al., 2009; Aguerre et al., 2011; Leytem et al., 2011). This strong diel trend in emissions can be associated with wind speed and temperature, as winds tend to be light in the late evening and early morning and then, in most instances, steadily increase throughout the day to reach a peak in the late afternoon. Temperature also increases from early morning to late afternoon, and then decreases again. Additionally, cattle activity tends to increase from morning to late afternoon as animals wake and begin to eat, drink, ruminate, defecate, and urinate. As these activities increase, one would expect an increase in CH_4 (and NH_3) emissions. There are also seasonal trends in emissions, the most prominent being in NH_3 emissions, with the lowest rates in winter compared with the other seasons (Amon et al., 2001; Powell et al., 2008; Bjorneberg et al., 2009; Flesch et al., 2009; Aguerre et al., 2011; Leytem et al., 2011). Powell et al. (2008), Flesch et al. (2009), and Aguerre et al. (2011) reported that barn emissions of NH_3 in Wisconsin were lowest in winter, with winter rates about one-half to one-third lower than those in the spring and summer, which was

Ammonia Emissions in Dairy Cattle Housing

As mentioned earlier, ammonia is not a greenhouse gas, however, ammonia emissions are estimated as part of the nitrogen balance approach. Emissions of NH_3 from dairy cattle housing systems have been strongly linked to dietary nitrogen intake, as this affects the amount of urea nitrogen excreted in urine. Of the nitrogen in the total crude protein (CP) typically consumed by a dairy cow on commercial dairy farms, 20 to 35 percent is secreted in milk and the remaining nitrogen from CP is excreted about equally in feces and urine. Feed nitrogen (N=CP÷6.25) use efficiency (percentage of feed nitrogen secreted as milk nitrogen) and the 50:50 fecal nitrogen:urinary nitrogen excretion ratio can be influenced greatly, however, by what is fed to the cow. Feeding nitrogen in excess of nutritional requirements has very few significant impacts on milk production or quality; it decreases feed nitrogen use efficiency and increases the relative amount of urea nitrogen excreted in urine. The urea nitrogen contained in cow urine (which is 55 to 80 percent of the nitrogen contained in urine, depending on concentrations of CP in the ration) is the major source of NH_3 emission from dairy farms. Urea is produced when nitrogen-rich proteins and/or non-protein nitrogen sources break down (mainly in the cow rumen), forming NH_3 gas that may be used by ruminal microbes to produce microbial proteins or can be absorbed through the ruminal wall to the blood stream. In the kidney, blood NH_3 from the digestive tract or tissue metabolism is eventually converted to urea before being excreted in the urine. Urease enzymes, which are present in feces and soil, rapidly convert excreted urea to ammonium, which can be hydrolyzed quickly into NH_3 gas and lost to the atmosphere. Thus, the increase in urea nitrogen excretion due to excessive ration CP increases NH_3 emissions during the collection, storage, and land application of manure (Rotz, 2004; Misselbrook et al., 2005; Powell et al., 2008; Arriaga et al., 2010).

Paul et al. (1998) examined the effects of altering dietary CP on NH_3 losses from dairy cows. They reported that NH_3 emissions during the first 24 hours following manure excretion were 38 and 23 percent of the total manure nitrogen from diets with 16.4 and 12.3 percent CP concentrations, respectively, and 22 and 15 percent of total manure nitrogen from diets containing 18.3 and 15.3 percent dietary CP, respectively. Misselbrook et al. (Misselbrook et al., 2005) reported that reducing dietary CP content resulted in less total nitrogen excretion and a smaller proportion of the excreted nitrogen being present in urine; urine nitrogen concentration was 90 percent greater for the high-CP than the low-CP diet.

However, Li et al. (2009) found no effect of lowering dietary CP in lactating dairy cattle on NH_3 emissions from the floor of a naturally ventilated freestall dairy barn at low and moderate temperatures (0 to 20°C). This lack of response to CP is likely due to the fact that urease activity is negligible at temperatures below 10°C (Bluteau et al., 2009). Factors that are essential in determining NH_3 emissions are manure or urine pH and the total ammoniacal nitrogen content, both of which are related to the dietary CP level.

The majority of NH_3 emissions from housing systems are due to the volatilization of NH_3 from urine deposition. As discussed above, nitrogen intake drives the amount of urea that is excreted in the urine. As this urine is deposited on barn floors, pastures, or dry-lots, it mixes with urease from either feces or soil and is then hydrolyzed to ammonium and, via effects of pH, converted to NH_3 and lost to the atmosphere. The loss of NH_3 happens rapidly, with most NH_3 losses occurring within 24 hours following deposition. Therefore, estimation of NH_3 emissions needs to take into account the amount of urea generated by the cow, pH (urine, manure, or soil), temperature, and air flow over the source. Strategies that reduce nitrogen excretion will be very beneficial in reducing NH_3 emissions from housing/pasture systems.

attributed to cold winter temperatures. In general, N_2O emissions from housing were found to be low and showed no discernible diel or seasonal trends (Bjorneberg et al., 2009; Ngwabie et al., 2009; Adviento-Borbe et al., 2010; Leytem et al., 2011), suggesting that these emissions from this sector of the production system are of relatively little concern. There are consistent reports of both diel and seasonal variations in both CH_4 and NH_3 emissions, so it is imperative that these factors be captured in any estimation of emissions for a given production system.

Emissions of CH_4 are dominated by enteric fermentation in housing/pasture systems. Amon et al. (2001) examined CH_4 emissions from a tie-stall dairy barn in Austria using either a slurry-based system or straw-based system. In both systems, about 80 percent of the net CH_4 emissions were due to enteric fermentation, with the remaining amount coming from the manure. Sun et al. (2008) measured CH_4 emission from dairy cows and fresh manure in chambers, and reported that fresh manure alone did not produce noticeable CH_4 fluxes. In some dairy production systems, manure is removed from the animal housing area frequently; therefore, CH_4 emissions from animal housing areas of a dairy can be largely attributed to enteric emissions.

N_2O emissions tend to be negligible from both animals and fresh manure. The majority of N_2O emissions result from manure storage, pasture, and land application of manures. Therefore, the main sources of N_2O emissions from animal housing would be from dry-lot dairies and stand-off pads, because there is potential for deposited nitrogen to be nitrified and denitrified under wet conditions and lost as N_2O. Luo and Saggar (2008) measured N_2O and CH_4 emissions from a dairy farm stand-off pad in New Zealand and reported N_2O fluxes from 0 to 3 g N_2O-N day^{-1}, which they attributed to the concentrations of water and nitrate in the pad materials. Overall, only 54 g of N_2O-N was emitted from the pad over the time of use, representing ~0.01 percent of the excreta nitrogen deposited on the pad.

While there have been overall improvements in milk production with breeding programs, there is no evidence that any breed of dairy cow produces less enteric CH_4. Münger and Kreuzer (Münger and Kreuzer, 2008) measured enteric CH_4 production from Holstein, Simmental, and Jersey cows and found no persistent differences in CH_4 yields, with average enteric CH_4 being approximately 25g CH_4 kg DMI^{-1}.

5.3.1.1 Method for Estimating Emissions from Dairy Production Systems

Method for Estimating CH_4 Emissions from Enteric Fermentation in Dairy Cows

- Mills et al. (2003) developed a series of submodels to estimate enteric CH_4 emissions from dairy and beef cattle. The optimal model appeared to be a nonlinear Mits3 equation, which is utilized by the DairyGEM Model (a subset of IFSM) (Rotz et al., 2011b) and is shown in Equation 5-1 (Mits3 equation) is based primarily on metabolizable energy intake, acid detergent fiber (ADF), and starch content of diet.
- Data sources are user input on dietary intake, as well as dietary data from the Feedstuffs Composition Table (Ewan, 1989; Preston, 2013).
- Use of the DairyGEM/Mits3 equation is recommended over the IPCC Tier 2 equation (IPCC, 2006) because it has proven to be more accurate, in general, for dairy cows.

Equation 5-1: Non-Linear Mits3 Equation

$$CH_4 = (E_{max} - [E_{max} \exp^{-cx}]) \times 0.018$$

Where:

CH_4 = Enteric methane emissions per day (kg CH_4 head^{-1} day^{-1})

E_{max} = Maximum possible CH_4 emissions (MJ head^{-1} day^{-1})

c = Shape parameter determining emission change with increasing metabolizable energy intake (see Equation 5-2)

x = Metabolizable energy intake (MJ head^{-1} day^{-1})

0.018 = Conversion of MJ to kg of CH_4 (kg CH_4 MJ^{-1})

The E_{max} is constant for all animals at 45.98 MJ/head/day. The shape parameter "c" is calculated from the dietary non-fiber carbohydrate (NFC) to acid detergent fiber (ADF) ratio in Equation 5-2.

Equation 5-2: Calculating Shape Parameter

$$c = -0.0011 \times \frac{NFC}{ADF} + 0.0045$$

Where:

c = Shape parameter determining emission change with increasing metabolizable energy intake (unitless)

NFC = [(100-NDF + CP + EE)/100] x DMI (kg head^{-1} day^{-1})

DMI = Dry matter intake (kg dry feed animal^{-1} day^{-1})

ADF = Acid Detergent Fiber (kg head^{-1} day^{-1})

NDF = Neutral Detergent Fiber (%)

CP = Crude Protein (%)

EE = Ether extract (%)

Mills et al. (2003) noted that nonlinear models have two advantages over linear models: 1) a maximum emission is set; and 2) it is explainable from a biological sense. The feedstuff characteristics needed to calculate emissions from dairy cattle are included in the example below (Ewan, 1989; Preston, 2013). The full table can be found in Appendix 5-B.

Table 5-6: Example Feedstuffs Table[a]

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)*	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Alfalfa Cubes	x91	57	57	25	57		18	30	29	36	46	40	2.0	11	1.30	0.23	1.9	0.37	0.33	20
Alfalfa dehydrated 17% CP	92	61	62	31	61	65.16	19	60	26	34	45	6	3.0	11	1.42	0.25	2.5	0.45	0.28	21
Alfalfa fresh	24	61	62	31	61	62.54	19	18	27	34	46	41	3.0	9	1.35	0.27	2.6	0.40	0.29	18

Source: Preston (2013).

[a] Column headings:

DM	= Dry matter	GE	= Gross energy	ASH	= Ash
TDN	= Total digestible nutrients	CP	=Crude protein	Ca	= Calcium
NEm	= Net energy for maintenance	UIP	= Undegradable intake protein	P	= Phosphorous
NEg	= Net energy for growth	CF	= Crude fiber	K	= Potassium
NEl	= Net energy for lactation	ADF	= Acid detergent fiber	Cl	= Chlorine
Mcal	= Megacalories	NDF	= Neutral detergent fiber	S	= Sulfur
cwt	= Centum weight (hundredweight)	eNDF	= effective neutral detergent fiber	Zn	= Zinc
DE	=Digestible energy	EE	= Ether extract	ppm	= parts per million

Method for Estimating Dairy Cows' GHG Emissions from Housing

Methane

- The DairyGEM Model (a subset of IFSM) (Rotz et al., 2011a) calculates CH_4 emissions from housing surfaces.
- DairyGEM uses the IPCC (2006) Tier 2 method to estimate CH_4 emissions when manure is allowed to accumulate in the housing.

Nitrous Oxide

- Nitrogen excreted estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O emissions from manure in housing.

Methane Emissions from Dairy Cows' Housing

The DairyGEM Model (Rotz et al., 2011a) calculates CH_4 emissions from barn floors using an empirical model developed from three freestall barns (Chianese et al., 2009c).

Equation 5-3: Calculating CH_4 Emissions from Barn Floors (Chianese et al., 2009c)

$$\mathbf{CH_4} = \mathbf{max(0.0, 0.13T)} \times \frac{\mathbf{A_{barn}}}{\mathbf{1000}}$$

Where:

CH_4 = Methane emissions per day (kg CH_4 head^{-1} day^{-1})

T = Barn temperature (˚C)

A_{barn} = Area of the barn floor covered with manure (m^2)

When manure is allowed to accumulate as a stockpile, on a dry-lot, or in a pit below the animal confinement, the DairyGEM model uses the IPCC (2006) Tier 2 method to estimate CH_4 emissions (Equation 5-4). This is the same equation used for estimating emissions from manure that is managed outside of housing (see Section 5.4.1 *Temporary Stack and Long-Term Stockpile* and 5.4.2 *Composting* for details).

Equation 5-4: IPCC Tier 2 Approach for Estimating CH_4 Emissions in Housing

$$E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{MCF}{100}$$

Where:

E_{CH4} = CH_4 emissions per day (kg CH_4 day^{-1})

m = Total dry manure per day [a] (kg dry manure day^{-1})

VS = Volatile solids (kg VS (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacity for manure (m^3 CH_4 (kg VS)$^{-1}$)

MCF = CH_4 conversion factor for the manure management system (%)

0.67 = Conversion factor of m^3 CH_4 to kg CH_4

The maximum CH_4 producing capacity (B_0) for manure varies by animal category and is provided in Table 5-19. The CH_4 conversion factors (MCF) for manure deposited on a dry-lot, stored in a deep pit, or from cattle bedding can be found in Table 5-7. The MCFs for manure stored as a stockpile are provided in Table 5-20 through Table 5-22. The MCFs for manure composted within housing are provided in Table 5-24.

Table 5-7: Methane Conversion Factors for Dry-Lots, Pit Storage Below Animal Confinement, and Cattle/Swine Bedding

Temperature		Dry-Lot	Pit Storage Below Animal Confinement and Cattle/Swine Deep Bedding	
			< 1 month	> 1 month
Cool	≤10 °C	1%	3%	17%
	11 °C			19%
	12 °C			20%
	13 °C			22%
	14 °C			25%
Temperate	15 °C	1.5%	3%	27%
	16 °C			29%
	17 °C			32%
	18 °C			35%
	19 °C			39%
	20 °C			42%
	21 °C			46%
	22 °C			50%
	23 °C			55%
	24 °C			60%
	25 °C			65%
Warm	26 °C	2%	30%	71%
	27 °C			78%
	≥28 °C			80%

Source: IPCC (2006).

The Sommer model is used to estimate emissions from any liquid manure (less than 10 percent dry matter) stored in housing. The estimation method for liquid manure can be found in Section 5.4.4 *Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks*.

Nitrous Oxide Emissions from Dairy Cows' Housing

To estimate nitrogen losses from housing, the amount of nitrogen excreted (N_{ex}) by each animal category is first estimated. Equation 5-5, Equation 5-6, and Equation 5-7 are the equations recommended by the American Society of Agricultural and Biological Engineers (ASABE) for estimating N_{ex}.

Equation 5-5: ASABE Approach for Estimating Nitrogen Excretion from Lactating Cows

$$\mathbf{N_{ex} = (Milk \times 2.303) + (DIM \times 0.159) + (DMI \times C_{CP} \times 70.138) + (BW \times 0.193) - 56.632}$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

Milk = Milk production per animal per day (kg milk animal^{-1} day^{-1})

DIM = Days in milk (days)

DMI = Dry matter intake (kg animal^{-1} day^{-1})

C_{CP} = Concentration of crude protein of total ration (g crude protein (g dry feed)$^{-1}$)

BW = Average live body weight (kg)

Equation 5-6: ASABE Approach for Estimating Nitrogen Excretion from Dry Cows

$$\mathbf{N_{ex} = (DMI \times 12.747) + (C_{CP} \times 1606.290) - 117.500}$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

DMI = Dry matter intake (kg dry feed animal^{-1}day^{-1})

C_{CP} = Concentration of crude protein of total ration (g crude protein (g dry feed) $^{-1}$)

Equation 5-7: ASABE Approach for Estimating Nitrogen Excretion from Heifers

$$\mathbf{N_{ex} = (DMI \times C_{CP} \times 78.390) + 51.350}$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

DMI = Dry matter intake (kg dry feed animal^{-1}day^{-1})

C_{CP} = Concentration of crude protein of total ration (g crude protein (g dry feed) $^{-1}$)

Some of the nitrogen excreted is volatilized as NH_3, hence, the estimation of NH_3 losses is necessary to estimate N_2O emissions using a nitrogen balance approach. The NH_3 lost from manure in housing is estimated as a fraction of N_{ex}, Koelsch and Stowell (2005) provide estimates on the typical NH_3 loss from different housing facilities and animal species as a fraction of N_{ex} (see Table 5-8). A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-8: Typical Ammonia Losses from Dairy Housing Facilities (Percent of N_{ex})

Facility Description	% Loss	Facility Description	% Loss
Open dirt lots (cool, humid region)	15 - 30	Roofed facility (shallow pit under floor)	10 - 20
Open dirt lots (hot, arid region)	30 - 45	Roofed facility (bedded pack)	20 - 40
Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	5 - 15	Roofed facility (deep pit under floor - includes storage loss)	30 - 40

Source: Koelsh and Stowell (2005).

N_2O is lost from the excreted nitrogen. A quantitative method for estimating N_2O emissions from solid manure is the IPCC Tier 2 approach, which is also used for the U.S. Greenhouse Gas Inventory (Equation 5-8). This estimation method is the same as the method present in the *Temporary Stack and Long-Term Stockpile* and the

Composting sections (See Sections 5.4.1 and 0). This equation will over-estimate the emissions from animal housing if some of the nitrogen excreted is managed outside of housing (i.e., the equation accounts for nitrogen loss due to NH_3 emissions but does not account for the quantity of nitrogen that is managed in manure management systems).

Equation 5-8: IPCC Tier 2 Approach for Estimating N_2O Emissions from Housing

$$\mathbf{E_{N_2O,housing} = n \times N_{ex} \times (1 - \%\ NH_3\ loss/100) \times EF_{N2O} \times \frac{44}{28} \times \frac{1}{1000}}$$

Where:

$E_{N2O,\ housing}$	= Nitrous oxide emissions from housing per day (kg N_2O day^{-1})
N	= Number of head of livestock species (animal)
N_{ex}	= Total nitrogen excretion per animal per day (g N $animal^{-1}$ day^{-1})
$\%NH_3$ loss	= Percent of N_{ex} lost as NH_3 in animal housing - see Table 5-8
EF_{N2O}	= N_2O emission factor for manure in housing (kg N_2O-N kg N^{-1})
$\frac{44}{28}$	= Conversion of N_2O-N emissions to N_2O emissions
$\frac{1}{1000}$	= Conversion of grams to kilograms

For manure in deep pits, dry-lots, or mixed with bedding, the emission factors are provided in Table 5-9. The N_2O emission factors for manure in housing that is stored in a stockpile are provided in Table 5-23. The emission factors for manure that is composted within a housing area are provided in Table 5-25.

Table 5-9: N_2O Emission Factors for Manure Stored in Housing

Category	N_2O Emission Factor (kg N_2O-N/ kg N)
Cattle and Swine Deep Bedding (Active Mix)	0.07
Cattle and Swine Deep Bedding (No Mix)	0.01
Pit Storage Below Animal Confinements	0.002
Dry-Lot	0.02

Source: IPCC (2006).

The remaining nitrogen excreted that is not lost as N_2O or volatilized as NH_3 in housing then enters manure storage and treatment. If data are not available to track the nitrogen that is transferred along with the manure-to-manure storage and treatment, the nitrogen can be estimated as described in Equation 5-9. However, this equation is overestimating the nitrogen transferring to manure storage and treatment as some nitrogen will be lost in housing. This remaining total nitrogen value is an input into the N_2O equations for manure stored or treated.

The DairyGEM Model provides daily estimates; users can refer to that model for a more in-depth analysis of their emissions.

Equation 5-9: Total Nitrogen Entering Manure Storage and Treatment

$$\mathbf{TN_{storage} = n \times N_{ex} \times (1 - \%\ NH_3\ loss/100) \times \frac{1}{1000}}$$

Where:

$TN_{storage}$	= Total nitrogen entering manure storage (kg N day^{-1})
N	= Number of head of livestock species (animal)
N_{ex}	= Total nitrogen excretion per animal per day (g N $animal^{-1}$ day^{-1})
$\%NH_3$ loss	= Percent of N_{ex} lost as NH_3 in animal housing - see Table 5-8
$\frac{1}{1000}$	= Conversion of grams to kilograms

5.3.1.2 Rationale for Selected Method for Estimating Emissions from Dairy Production Systems

There are a variety of methods and models available to estimate emissions from dairy production systems, ranging from simple carbon footprint models to highly complex process-based models for the determination of NH_3 and GHG emissions. The IPCC Tier 1 methodology provides a simplistic method used for country inventory purposes. When additional data are available, there are a series of equations that can be used to develop IPCC Tier 2 estimates. The data used for these estimates are typically easily obtainable from the production facility or available in a lookup table. While these methods provide estimates for emissions that may be suitable for a rough determination of emissions inventories, they are in some cases based on very limited data and may not be very representative of emissions at the farm level. The development of process-based models has provided a way to obtain a more detailed analysis of emissions at the farm scale.

A wide variety of models applicable to dairy production facilities were identified and evaluated, including: Carbon Accounting for Land Managers; Climate Friendly Food Carbon Calculator; Cool Farm Tool; CPLAN; DairyGEM; Dairy Wise; Farming Enterprise GHG Calculator; Farm GHG; Holos; Integrated Farm System Model (IFSM); Manure And Nutrient Reduction Estimator (MANURE); Manure DeNitrification-DeComposition (Manure DNDC); OVERSEER; and SIMS Dairy.

These models were evaluated to determine their suitability for use to determine emissions estimates for dairy production facilities in the United States. Eleven criteria were used to identify models that could be used to estimate CH_4 from enteric CH_4 production and CH_4, N_2O, and NH_3 from animal housing systems. Two of the criteria were considered critical: the model had to be relevant to U.S. climate and dairy production systems and it had to be publically available. If the models met these two criteria they were further ranked based on the remaining nine criteria. Four of the models considered met the critical criteria: DairyGEM, IFSM, Cool Farm Tool, and MANURE. Although DairyGEM is a subset of IFSM, it was included separately because DairyGEM only estimates emissions from the animal housing and manure storage area. Therefore, it is less cumbersome to use and requires fewer inputs.

Model Evaluation Criteria for Dairy Production Systems

1. The model is based on well-established, scientifically sound relationships among farm management inputs, emissions outputs (process-based/mass-balance model preferable).
2. The model is relevant to U.S. climate and dairy production systems.
3. The model can estimate CH_4, and N_2O, and NH_3 emissions from dairy housing systems (including enteric CH_4 production).
4. There is flexibility in the model to describe the production system (animals, feed, housing, and in-house manure management).
5. The model is easy to use and is designed to use easily obtainable farm information to determine emissions estimates.
6. Model emission estimates for both enteric CH_4 production and emissions associated with cattle housing are easily captured.
7. The model includes some mitigation strategies for reducing emissions and produces realistic changes in emissions values when these changes are made within the production system.
8. There is transparency in the model calculations, and technical guidelines are available to elaborate the methodologies used to obtain the emissions estimates.
9. The model has been tested/validated with on-farm data.
10. The model works reliably (little to no errors or program crashes).
11. The model is publicly available and accessible.

Out of these four models, DairyGEM had the most flexibility for describing the production system and met all of the specified criteria. In addition, this model implements emission estimate methodologies that are advanced beyond the IPCC Tier 2 determinations. It models CH_4 emissions from enteric fermentation and manure management and the nitrogen balance associated with nitrogen excreted in manure. The underlying methods in the DairyGEM model are recommended for determining CH_4 emissions from enteric fermentation and housing systems for dairy cattle (see further discussion in Appendix 5-E, Table 5-E-1, and subsequent relevant text). The estimates generated from this model could then be modified to account for mitigation strategies that could alter the emissions currently being generated on-farm. Some mitigation strategies are already embedded in the model, such as alternative feeding, manure handling/storage, and the use of bovine somatotropin, while others could be used by developing a table with modifiers based on literature values to determine how on-farm emissions could change with the implementation of these strategies. For N_2O emissions, a nitrogen balance approach (based on the concepts in DairyGEM) using nitrogen excretion equations from ASABE Standard D384.2 is recommended. The use of the ASABE equations takes into account the impact of dietary changes on nitrogen excretion.

5.3.2 Enteric Fermentation and Housing Emissions from Beef Production Systems

Because of differences in the diets, animal physiological state and age, and manure handling, the proportions and sources of GHGs differ among the cow-calf, stocker, and finishing segments of the beef cattle industry. A primary source of GHGs from the beef cattle industry is enteric CH_4, produced primarily in the rumen, although some CH_4 is also produced in the lower gut. In addition, CH_4 and N_2O may be produced from feces and urine on pastures and feedlot pen surfaces. Emissions from housing and manure handling (prior to entering a management system) are discussed, and equations for stockpiled manure (Section 5.4) can be applied for emission estimation.

Phetteplace et al. (2001) estimated GHG emissions from simulated beef and dairy[4] systems in the United States using modifications of the IPCC (1997) methodology. The systems were comprised of a base herd of mature cows plus calves and replacements, stocker calves, a feedlot, and a dairy with 100 lactating cows. They also evaluated emissions from calves that went through the entire cow-calf, stocker and feedlot system (cow-calf to feedlot). Greenhouse gas emissions head^{-1} (CO_2-eq) from Phetteplace et al. (2001) are presented in Table 5-10 (with the exception of the dairy herd).

Table 5-10: Simulated GHG Emissions for Ruminant Systems (kg CO_2-eq/head/year)

Item	Cow-calf	Stocker	Feedlot	Cow-calf Through Feedlot
Dietary TDN, %	62	57	88	62
GHG (kg CO_2-eq/head/year)				
Enteric CH_4	1,140	1,725	743	1,167
Manure CH_4	34	48	12	34
Total CH_4	1,175	1,773	755	1,201
N_2O	1,487	1,721	1,294	1,490
CO_2	127	380	1,245	252
Total CO_2-eq	2,788	3,874	3,294	2,944

Source: Phetteplace et al. (2001).

Elsewhere, Beauchemin et al. (2010) used the Holos model (Little et al., 2008) to conduct a life-cycle assessment of beef production in western Canada. Of total CO_2-eq, 63 percent was from enteric CH_4.[5] These are very similar to values reported by the U.S. Department of Agriculture (2004b). Sixty-one percent of CO_2-eq emissions were from the cow-calf herd, 19 percent were from replacement heifers, eight percent were from backgrounding operations, and 12 percent were from feedlots. Seventy nine percent of enteric CH_4 losses were from the cow herd, three percent from bulls, two percent from calves, seven percent from backgrounders, and nine percent from feedlots. N_2O contributions (CO_2-eq) as a percent of total GHG emissions were as follows: feedlot manure – two percent, feedlot soil – two percent, cow-calf herd soil – two percent, and cow-calf herd manure – 20 percent.

Cow-Calf and Bulls

There is no evidence that any breed of beef cow produces less enteric CH_4 than another. There are a few reports suggesting that efficient cattle (those selected for feed efficiency or residual feed intake (RFI)) may produce less enteric CH_4 (Nkrumah et al., 2006; Hegarty et al., 2007). However, Freetly and Brown-Brandl (2013) reported that cattle with greater feed efficiency actually produced more CH_4; thus raising some questions about the genetic factors associated with feed efficiency and CH_4 emissions. It is unclear whether the changes observed are a result of altered feed intake or are associated with a change in altered ruminal microbial population. Additionally, recent information indicates that there is an interaction between diet quality and feed efficiency on enteric CH_4 emissions, where efficient cows produce less CH_4 when grazing high-quality pasture but not when grazing poor-quality forage (Jones et al., 2011). Residual feed intake is moderately heritable—(0.28 to 0.58; Moore et al., 2009), thus it might be possible to genetically select for animals with lower enteric CH_4 production. An examination of the value for selection for low enteric CH_4 production has been conducted with sheep in New Zealand and Australia. Simulations using published data

[4] Discussion of emissions from dairy production systems can be found in Section 5.3.1.

[5] 5% of emissions were from manure CH_4, 23% from manure N_2O, 4% from soil N_2O, and 5% from energy CO_2.

indicate that without accurate feed intake information and a method by which many animals can be screened for CH_4 emissions, selection for lower enteric CH_4 production is not likely to be economically viable (Cottle et al., 2011).

Measurement of enteric CH_4 from grazing cattle has been conducted primarily from animals grazing improved pastures using micrometeorological methods and tracer techniques. Lassey (2007) summarized much of the CH_4 emissions data that had been collected using the SF_6 tracer technique. Intake was either calculated from a requirements model or from use of markers (Cr_2O_3 or Yb_2O_3). Estimated forage digestibility (in vitro) ranged from 48.7 to 83 percent, which resulted in estimated CH_4 conversion factors [i.e., enteric CH_4 as a percentage of gross energy intake (GEI)] ranging from 3.7 to 9.5 percent. The mean Ym from all of the studies was 6.25 and agrees reasonably well with that used by IPCC (2006) for cattle on pasture. Methane emissions from cows grazing improved pasture, Kentucky fescue, and Bermuda grass in the southern United States were reported by Pavao-Zuckerman et al. (1999) and DeRamus et al. (2003). In both of these studies significant reductions in enteric CH_4 unit^{-1} of animal weight gain resulted from the implementation of best management practices designed to improve pasture quality.

Enteric emissions estimates can be made using micrometeorological methods and tracer techniques. One report in which CH_4 emissions were measured from beef cows grazing native range in October and May illustrated a large variation in enteric emissions. In October, when cows were losing BW, they produced 87 g CH_4 head daily^{-1}, and on the same pasture in May they produced 252 g CH_4 head daily^{-1} (Olson et al., 2000). Westberg et al. (2001) measured CH_4 from cows grazing the same pasture across seasons and found similar results, with higher CH_4 emissions from cows grazing lush spring growth and the lowest emissions from grazing stockpiled fall pasture. These differences are attributable to differences in both DMI and forage quality. In general, as forage quality increases, DMI also increases. Some "rules of thumb" for DMI on pasture include the following:

- Poor quality pasture - DMI = 1 to 1.75 percent of body weight;
- Medium quality forage - DMI = 1.75 to 2.25 percent of body weight;
- High quality forage DMI = 2.25 to 3 percent of body weight.

Stockers

Enteric CH_4 emissions of stockers while grazing have been measured by Laubach et al. (2008), Tomkins et al. (2011), McGinn et al. (2011), and Boadi et al. (2002), using a variety of techniques including the SF_6 tracer, and several micrometeorological approaches. The same factors that affect CH_4 emissions from grazing beef cows are important in stocker cattle. Those factors are level of feed intake, digestibility of forage consumed, supplementation, and the chemical composition of the plants consumed. Enteric emissions estimates can be made using micrometeorological methods or, tracer techniques or can be predicted from IPCC Tier 2 methods (see enteric discussion). Critical variables include measurements or estimations of feed intake and feed quality (chemical composition). Many of the equations currently available may not accurately predict measured enteric emissions from grazing cattle (Tomkins et al., 2011).

Feedlot

Most estimates of enteric methane emission from finishing beef cattle are based on work using animals confined to respiration chambers, although a few studies have used micrometeorological methods in open feedlots. Enteric CH_4 losses from finishing beef cattle normally range from 50 to 200 L head^{-1} daily (Johnson and Johnson, 1995; McGinn et al., 2004; Beauchemin et al., 2008; Loh et al., 2008; Hales et al., 2012; 2013; Hales et al., 2014; Todd et al., 2014a; Todd et al., 2014b). In most studies in the U.S., diets have been based on DRC or SFC; whereas most studies in Canada the diets are based on barley. The IPCC Tier 2 (2006) enteric CH_4 conversion factor (Ym) for feedlot cattle is

3 ± 1 percent of GEI. There are few studies that have measured emissions of CH_4 and N_2O from feedlot pen surfaces and runoff control structures. The primary factors that control enteric methane emissions in feedlot cattle are feed intake, grain type, grain processing method, dietary roughage concentration and characteristics, and dietary fat concentration.

5.3.2.1 Method for Estimating Emissions from Beef Production Systems

Method for Estimating CH_4 Emissions from Enteric Fermentation in Beef Cattle

- IPCC Tier 2 approach, with some adjustment factors, based on diet, animal weight, pregnancy/lactation, activity (IPCC, 2006).
- Data sources are user inputs on dietary intake, lactation and pregnancy rates, animal weight, housing, and the Feedstuffs Composition Table (Ewan, 1989; Preston, 2013).
- Although the equations utilized are the same as existing inventory methods, the method takes into account a large database of feed types (found in Appendix 5-B, Feedstuff Composition Table), as well as reporting at the monthly, rather than annual, temporal scale.

Equation 5-10: IPCC Tier 2 Equation for Calculating Gross Energy Requirements for Beef Cattle

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

Where:

GE = Gross energy (MJ day^{-1})

NE_m = Net energy required by the animal for maintenance (MJ day^{-1})[a]

NE_a = Net energy for animal activity (MJ day^{-1})[b]

NE_l = Net energy for lactation (MJ day^{-1})[c]

NE_{work} = Net energy for work (MJ day^{-1})[d]

NE_p = Net energy required for pregnancy (MJ day^{-1})[e]

REM = Ratio of net energy available in a diet for maintenance to digestible energy consumed[f]

NE_g = Net energy needed for growth (MJ day^{-1})[g]

REG = Ratio of net energy available for growth in a diet to digestible energy consumed[h]

DE = Digestible energy expressed as a percent of gross energy (%)

[a] Calculated using Eqn10.3 in IPCC (2006) based on body weight ("BW").
[b] Calculated using Eqn 10.4 in IPCC (2006) based on NE_a and feeding situation.
[c] Calculated using Eqn 10.8 in IPCC (2006) based on milk production ("milk") and milk fat ("fat").
[d] Calculated using Eqn 10.11 in IPCC (2006) based on information on daily hours of work ("work").
[e] Calculated using Eqn 10.13 in IPCC (2006) based on NE_m and pregnancy status.
[f] Calculated using Eqn 10.14 in IPCC (2006) based on DE.
[g] Calculated using Eqn 10.13 in IPCC (2006) based on body weight, mature weight ("MW"), and daily weight gain ("WG").
[h] Calculated using Eqn 10.15 in IPCC (2006) based on DE.

Equation 5-11: IPCC Tier 2 Equation for Calculating Enteric CH_4 Emissions from Beef Cattle

$$\mathbf{DayEmit} = \frac{\mathbf{GE \times Y_m/100}}{\mathbf{55.65}}$$

Where:

DayEmit = Emission factor (kg CH_4 head^{-1} day^{-1})

GE = Gross energy intake (MJ head^{-1} day^{-1})

Y_m = CH_4 conversion factor, which is the fraction of GE in feed converted to CH_4 (%)

55.65 = A factor for the energy content of methane (MJ kg $CH_4$$^{-1}$)

The DE ultimately used in the IPCC Tier 2 equation (in Equation 5-11) will be weighted based on portion of total feed intake from a particular feed type. The DE data for particular feedstuffs can be found in Appendix 5-B. The recommended Ym for beef replacement heifers, steer stockers, heifer stockers, beef cows, and bulls is 6.5 percent for all regions of the country. For feedlot cattle, the IPCC (2006) Ym of 3 percent is adjusted based on diets. All feedlot cattle initially start with a baseline Ym of three percent (IPCC, 2006).The correction factors to Ym for feedlot cattle for different scenarios are provided below (see Appendix 5-A for additional details). The Ym used for calculating emissions for these cattle is modified based on animal diets, as indicated in Table 5-11.

Table 5-11: Determination of Adjusted Methane Conversion Factor (Ym) for Feedlot Cattle

Variable	Ym (as a % of GEI)
Baseline Ym (IPCC, 2006)	**3%**
Ionophore in diet (Tedeschi et al., 2003; Guan et al., 2006):	
▪ Yes	No change
▪ No	Increase Ym by 4% (Ym= 3% x 1.04= 3.12% of GEI)
Fat Content (Zinn and Shen, 1996; Beauchemin et al., 2008; Martin et al., 2010) (For each percent of added fat (as supplemental fat or in byproducts such as distillers grain that contain about 10 percent fat), decrease by four percent to a maximum of a 16 percent decrease)	
▪ 1% supplemental fat	Decrease Ym by 4% (Ym = 3% x 0.96 = 2.88%)
▪ 2% supplemental fat	Decrease Ym by 8% (Ym = 3% x 0.92 = 2.76%)
▪ Four or higher added fat content	Decrease Ym by 16% (Ym= 3% x 0.84=2.52%)
Grain Type (Beauchemin and McGinn, 2005; Archibeque et al., 2006; Hales et al., 2012):	
▪ Grain in animal diet is steam flaked (SF) or high moisture (HM)	No Change
▪ Grain in animal diet is unprocessed (UP) or dry rolled (DR)	Increase Ym 20% (Ym = 3% x 1.2 = 3.6%)
▪ Grain in diet is barley rather than corn or sorghum	Increase Ym 30% (Ym = 3% x 1.3 = 3.9)
Grain Concentration (see Appendix 5-A for details and references):	
▪ Diet contains more than 60 percent grain	No Change
▪ Diet contains 45 to 60 percent grain	Increase Ym 10% (Ym= 3% x 1.1 = 3.3%)
▪ Diet contains less than 45 percent grain	Increase Ym 40% (Ym = 3% x 1.40 = 4.2%)

Method for Estimating Beef Cattle GHG Emissions from Housing

Methane

- The IPCC (2006) Tier 2 method can be used to estimate CH_4 emissions when manure is allowed to accumulate on feedlot pen surfaces as described below.

Nitrous Oxide

- Nitrogen excreted estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O emissions from manure in housing.

Emissions from Feedlot Pen Surfaces

There are few, if any, studies that have measured CH_4 or N_2O emissions from beef cattle feedlot pen surfaces and retention ponds. The study of Todd et al. (2014a; 2014b) suggests there is little CH_4 production from feedlot pen surfaces. The use of the IPCC (2006) methodologies is recommended to estimate emissions from feedlot pens and retention ponds.

In order to estimate CH_4 emissions from beef feedlot pen surfaces, the quantity of volatile solids excreted is first estimated. These can be estimated by lab testing samples from the facility or using values from the ASABE Standard D384.2 (ASABE, 2005).[6] CH_4 emissions from the pen surface can be estimated using the IPCC (2006) Tier 2 approach as outlined in section 5.4.1.2. For cattle feedlots, a maximum CH_4 production capacity (B_0) of 0.33 m^3/ kg volatile solids is assumed (Table 5-19) and the CH_4 conversion factor for pen surfaces ranges from 1 to 2 percent of B_0, depending upon environmental temperature (Table 5-20). Once manure is scraped from the pens and removed, the methods described in section 5.4.1 can be used to estimate CH_4 emission from manure that is composted or stored in stockpiles.

In order to estimate N_2O emissions from the pen surfaces of beef feedlots the quantity of nitrogen excreted on to the pen surface must be known. This can be estimated using Equation 5-12 from the ASABE Standard D384.2. For a beef feedlot, a default value of 0.069 kg of N kg dry manure^{-1} can be used if N_{ex} is not calculated.

[6] Volatile solids values can be estimated from equations (1) or (2) in section 4.3.1 of ASABE D384.2. Default volatile solids values are also presented in Table 5-32 of this document.

Equation 5-12: ASABE Approach for Estimating Nitrogen Excretion from Beef Cattle

$$\mathbf{N_{ex}} = \sum_{\mathbf{x=1}}^{\mathbf{n}} \frac{\mathbf{DMI_x \times C_{CP-x} \times DOF_x}}{\mathbf{6.25}} - [\mathbf{41.2} \times (\mathbf{BW_F} - \mathbf{BW_I})]$$
$$+ \left[\mathbf{0.243 \times DOF_T} \times \left[\frac{\mathbf{BW_F + BW_I}}{\mathbf{2}}\right]^{\mathbf{0.75}} \times \left[\frac{\mathbf{SRW}}{\mathbf{BW_F \times 0.96}}\right]^{\mathbf{0.75}}\right.$$
$$\left. \times \left[\frac{\mathbf{BW_F - BW_I}}{\mathbf{DOF_T}}\right]^{\mathbf{1.097}}\right]$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

DMI_x = Dry Matter Intake for ration x (kg dry feed animal^{-1}day^{-1})

C_{CP-x} = Concentration of crude protein of total ration (g crude protein g dry feed^{-1})

DOF_x = Days on feed for an individual ration (days)

BW_F = Live body weight at finish of feeding period (kg)

BW_I = Live body weight at the start of feeding period (kg)

DOF_T = Total days on feed from start to finish of feeding periods (days)

SRW = Standard reference weight for expected final body fat (kg)

x = Ration number

n = Total number of rations fed

Some of the nitrogen excreted is volatilized as NH_3, hence, the estimation of NH_3 losses is necessary to estimate N_2O emissions using a nitrogen balance approach. The NH_3 lost from manure in housing is estimated as a fraction of N_{ex}. Koelsch and Stowell (2005) provide estimates on the typical NH_3 loss from different housing facilities as a fraction of N_{ex} (see Table 5-12). A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-12: Typical Ammonia Losses from Beef Cattle Housing Facilities Expressed as a Percent of N_{ex}

Facility Description	% Loss	Facility Description	% Loss
Open dirt lots (cool, humid region)	30 – 45	Roofed facility (bedded pack)	20 - 40
Open dirt lots (hot, arid region)	40 – 60	Roofed facility (deep pit under floor, including storage loss)	30 - 40

Source: Koelsh and Stowell (2005).

An alternative approach is to use the equation of Todd et al. (2013) which calculates monthly feedlot NH_3 emissions as a function of dietary crude protein and average monthly temperature (Equation 5-13).

Equation 5-13: Monthly Beef Feedlot NH_3 Emissions as a Function of Dietary Crude Protein and Monthly Temperature

$$\mathbf{Ln(NH_3) = 8.82 - 1629\left[\frac{1}{T}\right] + 0.108 \times CP}$$

Where:

NH_3 = NH_3 emission from housing per day (g NH_3 head^{-1} day^{-1})

T = Average monthly temperature (K)

CP = Dietary crude protein as a fraction of dry matter (%)

N_2O emissions are calculated using the IPCC (2006) Tier 2 method and dry-lot emission factors described in Equation 5-8 and Table 5-9. The quantity of nitrogen that leaves the feedlot pens in manure can then be calculated using Equation 5-9. N_2O-N losses from manure collected and removed from the pens can be determined from manure nitrogen using Equation 5-27 and Equation 5-29 and the emission factors in Table 5-23 and Table 5-25 found in Section 5.4 Manure Management. NH_3 losses from manure nitrogen removed from the pens can be calculated as described in Appendix 5-C.1 and 5-C.3.

5.3.2.2 Rationale for Selected Method for Estimating Emissions from Beef Production Systems

Cow-Calf, Bulls, and Stockers

The most appropriate predictions available for entity scale estimation are IPCC Tier 2 methods for grazing cattle. Critical variables that are important to define in order to generate prediction methods include measurements or estimations of feed intake and feed quality (chemical composition) for pasture or rangelands. If the intake is not known, intake prediction equations/models such as NRC (2000) can be used. The NRC (2000) provides an equation for the calculation of DMI for grazing beef cows and for stocker cattle: NEm intake = $SBW^{0.75}$ * (0.04997 * NEm^2 + 0.04631) where NEm is the estimated Mcal kg^{-1} of the pasture, and SBW is the average shrunk body weight for the period of grazing (kg). The requirement for knowledge of the NEm concentration of the pasture may limit the usefulness of the prediction in some situations.

In situations in which the herd is housed in a dry-lot or barn facility, emission factors for CH_4 and N_2O associated with pen surfaces, manure storage, and animal movement/manure disturbance would be appropriate.

Feedlot

Ellis et al., (2009) reported that several equations appeared to be good predictors of enteric CH_4 losses from feedlot cattle based on Canadian studies. However, many of those equations tend to greatly overestimate enteric losses when compared with data from cattle fed a typical southern plains finishing diet (Hales et al., 2012; 2013; Todd et al., 2014a; Todd et al., 2014b). Although Kebreab et al. (2008) reported that MOLLY and IPCC Tier 2 (2006) gave predicted values similar to measured values with feedlot cattle, there was a large variability in individual animals with errors of 75 percent or greater. Kebreab et al. (2008) noted the average Ym (MJ enteric CH_4 MJ GEI^{-1}) for feedlot cattle based on experimental data was 3.88 percent of GEI (range 3.36 to 4.56), which was higher than the IPCC (2006) value of 3.0 percent and the recently obtained values with typical finishing diets of 2.85 percent (Hales et al., 2012; 2013).

Currently, IPCC Tier 2 may be the most useful methodology for prediction of enteric emissions from feedlot beef cattle. Unfortunately, the Tier 2 method does not allow for estimating changes in enteric emissions related to changes in diet or management.

Therefore, a modified IPCC (2006) method is recommended to estimate enteric CH_4 emissions from beef cattle fed high concentrate finishing diets. The CH_4 conversion factor (Ym) will be adjusted by factors in the animals' diets as described in Section 5.3.2.1. A baseline scenario based on typical U.S. beef cattle feeding conditions is established, and the Ym values are adjusted based on published research. Emission values are modified using correction factors that are based on changes in animal management and feeding conditions from the baseline scenario.

5.3.3 Enteric Fermentation and Housing Emissions from Sheep

GHG emissions associated with sheep production include enteric CH_4 emissions, manure and bedding emissions, and emissions associated with grazing and manure application to land.

The New Zealand Ministry for the Environment (2010) estimated that sheep younger than a year of age emit 5.1 percent of GEI as enteric CH_4, and adult sheep emit 6.3 percent of their GEI as CH_4. These emission factors, when combined with population estimates, result in baseline enteric emissions of 11.60 kg CH_4 head^{-1} year^{-1}. Sheep are also estimated to deposit 15.9 kg N head^{-1} year^{-1}.

Lassey (2007) summarized the enteric emissions measurements from grazing sheep trials from New Zealand and Australia in which the SF_6 tracer technique was used. Forage characteristics ranged from lush (in vitro digestibility estimate of 82 percent) to poor quality (called "dead," with an in vitro digestibility of 54 percent). Intake was measured using complete fecal collection or a marker (n-alkane). Enteric CH_4 emissions ranged from 11.7 g day^{-1} for sheep fed forage of higher quality (6.9 percent of GEI) to 35.2 g day^{-1} for sheep fed forage of lower quality (6.3 percent of GEI). The average enteric emissions were 5.39 percent of GEI, or 23.5 g day^{-1}. In general, lower forage quality resulted in a greater amount of CH_4 emitted as a proportion of the energy intake than did higher forage quality.

New Zealand pastures grazed by sheep had elevated N_2O emissions (7.4 g N_2O-N ha^{-1} day^{-1} vs. 3.4g N_2O-N ha^{-1} day^{-1}) compared with control, but significantly less than that observed when cattle grazed (32.0 g N_2O-N ha^{-1} day^{-1}) (Saggar et al., 2007). The data were used to evaluate the NZ-DNDC model, a process-based New Zealand whole farm model. To our knowledge there are no published estimates of GHG emission from sheep manure systems.

5.3.3.1 Method for Estimating Emissions from Sheep

Method for Estimating Enteric Fermentation CH_4 Emissions from Sheep

- Howden equation (Howden et al., 1994), based on dietary DMI.
- The equation from Howden et al. (1994) estimates emissions based solely on DMI; hence, emission factors not utilized.

Equation 5-14: Equation for Enteric Fermentation Emissions from Sheep (Howden et al., 1994)

$$\mathbf{CH_4 = Intake \times 0.0188 + 0.00158}$$

Where:

CH_4 = Methane emissions (kg CH_4 head^{-1} day^{-1})

Intake = Dry Matter Intake (kg head^{-1} day^{-1})

The dry matter data for particular feedstuffs can be obtained from Appendix 5-B.

No emissions estimation methods have been provided for housing as most sheep are kept on pasture and minimal emissions are expected.

5.3.3.2 Rationale for Selecting Method for Estimating Emissions from Sheep

Howden et al. (1994) generated an equation from which to predict CH_4 emissions from sheep. Equation 5-7 resulted from a linear extrapolation of DMI to emissions. It has since been evaluated and found to be robust enough to be the equation used in the Australian National Greenhouse Gas Inventory. Klein and Wright (2006) measured CH_4 from sheep in respiration chambers and compared their results to the Howden et al. (1994) equation. Actual CH_4 averaged 1.1 g head^{-1} (SE ± 0.05) and predicted CH_4 was 1.1 g head^{-1} (SE ± 0.02). A potential concern regarding the Howden equation is that much of the data included in the analysis was based on tropical forages. Nonetheless, when intake data are available, the Howden equation presents the best method by which to estimate sheep enteric emissions. When intake is not available, the IPCC Tier 2 method of estimation should be used. Emissions from feedlot sheep should use the Ym values from Blaxter and Clapperton's original paper (1965) in which they measured CH_4 emissions from sheep with respiration calorimetry chambers. Sheep fed highly digestible diets at three times maintenance produced 35 percent less CH_4 (kcal 100K kcal of feed energy^{-1}) than those fed similar diets at maintenance; thus, a reduced Ym value is warranted. The equation is CH_4 = 1.3 + [0.112 × (% digestibility/100)] + [ME intake/maintenance ME requirement] × [2.37 - 0.050 × (%digestibility/100)].

5.3.4 Enteric Fermentation and Housing Emissions from Swine Production Systems

Sources of GHG emissions include enteric fermentation; manure stored within the animal housing, whether it is stored as a liquid or mixed with bedding; emissions that occur during the transport of manure to an external manure storage structure; the outside manure storage structure; emissions that occur during transport of manure to the field; and emissions following land application of manure. Because GHG mitigation has not been a focus of U.S. research for the swine industry nor a high priority for swine producers, data are not readily available to identify the magnitude of each of the above points of emission within a farm. However, emissions of CH_4 are expected to occur primarily during manure storage, and emissions of N_2O are expected to predominate following land application of manure.[7] Often manure is stored underneath the pig housing in a deep pit. For this reason, emissions discussion in this section includes in-house manure storage and comparison of in-house manure storage systems with systems that store manure externally. Because swine feeds are dry, emissions of GHG from feed storage areas are believed to be negligible.

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[7] Greenhouse gas emissions resulting following land application are addressed separately in the sections on Chapter 3: Croplands and Grazing Lands.

Greenhouse gas emission data from swine facilities is somewhat limited. Liu et al. (2011a) reported that grow/finish pigs emitted 42 to 79 mg CH_4 kg BW^{-1} daily from chambers where pigs were housed with manure. Daily emissions of N_2O ranged from 11.4 to 12.4 mg N_2O kg BW^{-1} (Li et al., 2011). These values are somewhat higher than data used by Verge et al. (2009) in calculating GHG emissions from Canadian pork production (43 mg CH_4 kg BW^{-1} and 4 mg N_2O kg BW^{-1}). Philippe et al. (2007) observed GHG emissions in the range reported by Li et al. (2011) though their observations were in European deep litter and slatted floor systems. The reported gaseous emissions from pigs raised on the slatted floor and on the deep litter were, respectively, 0.54 and 1.11 g pig^{-1} day^{-1} for N_2O, and 16.3 and 16.0 g pig^{-1} day^{-1} for CH_4.

Liu et al. (2011a) conducted a meta-analysis to identify factors that contribute to GHG emissions from swine production. Findings, shown in Table 5-13, illustrate that type of emission source (swine buildings or manure storage facilities) was not significant for CH_4 and N_2O emissions. Swine category (stage of production) and geographic location was significant for both of the GHG gases. Neither temperature nor size of operation was significant in the overall analysis.

Table 5-13: P Values of Main Effects on GHG Emissions from Swine Operations

Cause of Variation	CH_4 (n=76)	N_2O (n=53)
Emission source	0.94	0.93
Swine category	0.05	<0.01
Geographic region	0.04	0.02
Temperature	0.20	0.95
Size of operation	0.89	0.24

Source: Liu et al. (2011a).

Within the meta-analysis, Liu et al. (2011a) found that swine buildings with straw-flow systems generated the lowest CH_4 and N_2O emissions of systems compared, while pit systems generated the highest CH_4 emissions, and bedding systems generated the highest N_2O emissions. Emissions from lagoons and slurry storage basin/tanks were compared; lagoons generated significantly higher N_2O emissions than slurry storage basin/tanks, while CH_4 emissions were not different. Straw-based bedding resulted in numerically higher CH_4 but lower N_2O emissions when compared with sawdust or corn stalk bedding systems. Liu et al. (2011a) observed an increasing trend for CH_4 emissions as manure removal frequency decreased ($P = 0.13$). Deep pits and pits flushed using lagoon effluent also generated relatively high CH_4 emissions. Results for N_2O emissions showed very high uncertainties ($P = 0.49$). Deep pits and pits with manure removed every three or four months had relatively higher N_2O emissions. A summary of other findings from the meta-analysis conducted by Liu et al. (2011a) showed that CH_4 emissions from slurry storage facilities without covers were significantly higher than from those with covers.

The highest CH_4 emissions were from farrowing swine, and were significantly higher than those from finishing and nursery swine. Compared with farrowing swine, the gestating swine had significantly lower CH_4 emissions. The highest N_2O emissions were from gestating swine and were significantly higher than those from finishing swine.

North American studies reported significantly higher CH_4 emissions from swine operations than European and Asian studies (Liu et al., 2011a) . This is probably due to the different prevailing manure handling systems and different manure handling practices in different regions. Emissions of CH_4 from lagoons and manure storage facilities increased with increasing temperature. For swine buildings, temperature was not a significant factor.

5.3.4.1 Method for Estimating Emissions from Swine Production Systems

Method for Estimating Enteric Fermentation CH_4 Emissions from Swine

- IPCC Tier 1 approach, using U.S. emission factor of 1.5 kg CH_4/head/year. (IPCC, 2006).
- Sole data source is the IPCC Tier I emission factor for swine. User input is total number of head, regardless of class or weight.

Equation 5-15: Equation for Enteric Fermentation Emissions from Swine (IPCC, 2006)

$$\mathbf{CH_4 = Population \times 0.00411}$$

Where:

CH_4	= Methane emissions per day (kg CH_4 day^{-1})
Population	= Number of swine (head)
0.00411	= Daily CH_4 emissions from each animal (kg $head^{-1}$ day^{-1})

Method for Estimating Swine GHG Emissions from Housing

Methane

- The IPCC (2006) Tier 2 method is used to estimate CH_4 emissions when manure is allowed to accumulate below the animal confinement as described below.

Nitrous Oxide

- Nitrogen intake, retention, and excretion estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O from manure in housing.

Methane Emissions from Swine Housing

The IPCC (2006) Tier 2 equation is used to estimate CH_4 emissions when manure is allowed to accumulate in a pit below the animal confinement. The estimation method is provided in Equation 5-4. The maximum CH_4 producing capacity for swine is provided in Table 5-19. The MCFs for manure stored in a deep pit or from swine bedding is provided in Table 5-7.

Nitrous Oxide Emissions from Swine Housing

To estimate nitrogen losses from swine housing, the amount of nitrogen excreted (N_{ex}) for each animal classes are first estimated. Equation 5-16 describes the relationship between nitrogen intake, retention, and excretion for swine. Equation 5-17, Equation 5-18, Equation 5-19, and Equation 5-20 provide the methods for estimating the nitrogen intake and retention for the different swine classes as recommended by the ASABE.

Equation 5-16: ASABE Approach for Estimating Nitrogen Excretion from Swine

$$\mathbf{N_{ex} = N_{intake} - N_{Retention}}$$

Where:

N_{ex} = Total nitrogen excretion per animal (g $animal^{-1}$)

N_{intake} = Nitrogen intake per finished animal (g $animal^{-1}$)

$N_{retention}$ = Nitrogen retained per finished animal (g $animal^{-1}$)

Equation 5-17: ASABE Approach for Estimating Nitrogen Excretion from Grow-Finish Pigs

$$\mathbf{N_{intake} = ADFI_G \times C_{CP} \times \frac{DOF_G}{625}}$$

$$\mathbf{N_{retention} = \frac{BW_F \times DP_F \times FFLP_F}{159.4} - BW_I \times [DP_F - 0.05 \times (BW_F - BW_I)] \times \frac{FFLP_F + 0.07 \times (BW_F - BW_I)}{159.4}}$$

Where:

N_{intake} = Nitrogen intake per finished animal (g $animal^{-1}$)

$N_{retention}$ = Nitrogen retained per finished animal (g $animal^{-1}$)

$ADFI_G$ = Average daily feed intake over finishing period (g day^{-1})

C_{CP} = Concentration of crude protein of total (wet) ration (%)

DOF_G = Days on feed to finish animal (grow-finish phase) (days)

BW_F = Final (market) body weight (kg)

DP_F = Average dressing percent (yield) at final weight (%)

BW_I = Initial body weight (kg)

$FFLP_F$ = Average fat-free lean percentage at final weight (%)

Equation 5-18: ASABE Approach for Estimating Nitrogen Excretion from Weaning Pigs

$$\mathbf{N_{intake}} = \mathbf{ADFI_G} \times \mathbf{C_{CP}} \times \frac{\mathbf{DOF_N}}{\mathbf{625}}$$

$$\mathbf{N_{retention}} = \mathbf{DOF_N} \times \mathbf{FFLG_G} \times \frac{\mathbf{1 + [0.137 \times (BW_{F-N} + BW_{I-N})]}}{\mathbf{125.8}}$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_G$ = Average daily feed intake over finishing period (g day^{-1})

C_{CP} = Concentration of crude protein of total (wet) ration (%)

DOF_N = Days on feed to finish animal (nursery phase) (days)

$FFLG_G$ = Average fat-free lean gain from 20 to 120kg (g day^{-1}) [a]

BW_{F-N} = Final body weight in nursery phase (kg)

BW_{I-N} = Initial body weight in nursery phase (kg)

[a] Recommended values are: 350 g day^{-1} for high lean growth capacity pigs; 325 g day^{-1} for high-moderate lean growth capacity pigs; and 300 g day^{-1} for moderate lean growth capacity pigs.

Equation 5-19: ASABE Approach for Estimating Nitrogen Excretion from Gestating Sows

$$\mathbf{N_{intake}} = \mathbf{ADFI_S} \times \mathbf{C_{CP}} \times \left(\frac{\mathbf{GL}}{\mathbf{625}}\right)$$

$$\mathbf{N_{Retention}} = (\mathbf{GLTG} \times \mathbf{36.8}) + (\mathbf{LITTER} \times \mathbf{39.1})$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_S$ = Average daily feed intake during gestation (g day^{-1})

C_{CP} = Concentration of crude protein (%)

GL = Gestation period length (days)[a]

GLTG = Gestation lean tissue gain (kg)[b]

LITTER = Number of pigs in litter (head)

[a] Assumed to be 115 days.
[b] Recommended value from ASABE is 19.205 kg.

Equation 5-20: ASABE Approach for Estimating Nitrogen Excretion from Lactating Sows

$$\mathbf{N_{intake}} = \mathbf{ADFI_{LACT}} \times \mathbf{C_{CP}} \times \left(\frac{\mathbf{LL}}{\mathbf{625}}\right)$$

$$\mathbf{N_{Retention}} = (\mathbf{38.6} \times \mathbf{LLTG}) + (\mathbf{LW_{WEAN}} \times \mathbf{32}) - (\mathbf{LW_{BIRTH}} \times \mathbf{36.8})$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_{LACT}$ = Average daily feed intake during lactation (g day^{-1})

C_{CP} = Concentration of crude protein (%)

LL = Lactation length (days to weaning) (days)

LLTG = Lactation lean tissue gain (kg)[a]

L_{WEAN} = Litter weight at weaning (kg)

LW_{BIRTH} = Litter weight at birth (kg)

[a] Recommended value from ASABE is -4.20 kg.

Some of the nitrogen excreted is volatilized as NH_3, hence, the estimation of NH_3 losses is necessary to estimate N_2O emissions using a nitrogen balance approach. The NH_3 lost from manure in housing is estimated as a fraction of N_{ex} according to the ranges provided in Table 5-14. A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-14: Typical Ammonia Losses from Swine Housing Facilities (Percent of N_{ex})

Facility Description	% Loss	Facility Description	% Loss
Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	5 - 15	Roofed facility (bedded pack)	20 - 40
Roofed facility (shallow pit under floor)	10 - 20	Roofed facility (deep pit under floor - includes storage loss)	30 - 40

Source: Koelsh and Stowell (2005).

The IPCC (2006) Tier 2 approach is used for N_2O emissions from manure stored in housing. The estimation method is provided in Equation 5-8. The N_2O emission factors can be found in Table 5-9.

The remaining nitrogen excreted that is not lost as N_2O or volatilized as NH_3 in housing then enters manure storage and treatment. If data are not available to track the nitrogen that is transferred along with the manure to manure storage and treatment, the nitrogen can be estimated as described in Equation 5-9. However, this equation is overestimating the nitrogen transferring to manure storage and treatment as some nitrogen will be lost in housing. This remaining total nitrogen value is an input into the N_2O equations for manure stored or treated.

N_2O-N losses from manure collected and removed from housing can be determined from manure nitrogen using equations from Section 5.4 Manure Management for the appropriate manure management system. NH_3 losses from manure nitrogen removed from housing can be calculated using the methodology presented in Appendix 5-C.1 and 5-C.3.

5.3.4.2 Rationale for Selecting Method for Estimating Emissions from Swine

Miles et al. (2006) suggest that a robust model for enteric and housing emissions would include factors such as house management, animal size and age, pH, and manure moisture. Due to the current data limitations, an NH_3 and GHG estimation model should minimally include number of animals, excreta moisture content, diet protein and fiber content, and excreta pH. The challenge is that these criteria may not be readily available to the farm manager.

Liu et al. (2011a) compared literature values with IPCC values and concluded that the variation of the measured CH_4 and N_2O housing emission rates has not been adequately captured by the IPCC approaches. For CH_4 emissions, the differences between the IPCC-estimated emission rates and measured values were significantly influenced by type of emission source, geographic region, and measurement methods. Larger differences between estimated and measured CH_4 emission rates were observed in North American studies than in European studies. In North American studies, the results of meta-analysis indicated an overestimation by the IPCC approaches for CH_4 emissions from lagoons (pooled relative difference: -33.9%; 95% CI: -66.8% to -0.01%), and the discrepancy between the IPCC-estimated emissions and the measured values occurred mainly at lower temperatures. In European studies, the results indicated an overestimation of the IPCC approaches in swine buildings with pit systems. For N_2O emissions from swine operations, an overall underestimation of the IPCC approaches was observed in European studies but not in North American studies. In European studies, the pooled N_2O emission factors for swine buildings with pit systems was 1.6% (95% CI, 0.6% to 2.7%), while the IPCC default emission factor for pit systems is 0.2%. Larger uncertainties were observed for measured N_2O emissions from bedding systems and from straw flow systems.

Model Evaluation Criteria for Swine Production Systems

1. The model is based on well-established scientifically sound relationships between farm management inputs and emissions outputs (process-based model or mass-balance model preferable);
2. The model is relevant to U.S. climate and swine production systems;
3. The model can estimate CH_4, N_2O, and NH_3 emissions from enteric fermentation and swine housing systems;
4. There is flexibility in the model to describe the production system (animals, feed, housing, and in-house manure management);
5. The model is easy to use and is designed to use easily obtainable farm information to determine emissions estimates;
6. The model includes some mitigation strategies for reducing emissions, and produces realistic changes in emissions values when these changes are made within the production system;
7. There is transparency in the model calculations, and technical guidelines are available to elaborate the methodologies used to obtain the emissions estimates;
8. The model has been tested/validated with on-farm data;
9. The model works reliably (little to no errors or program crashes); and
10. The model is publicly available and accessible.

In order to consider an alternative to the IPCC approach, a wide variety of models applicable to swine production facilities were identified and evaluated, including 1) CAR Livestock, 2) Manure And Nutrient Reduction Estimator (MANURE), 3) COOL Farm Tool, 4) Carbon Accounting for Land

Managers, 5) Farming Enterprise GHG Calculator, 6) CPLAN, and 7) Holos. These models were evaluated by 10 criteria (see box) to determine their suitability for use in determining emissions estimates for swine production facilities in the United States. Two of these criteria were considered to be critical, in that if they were not met by the model, they could not be considered for use (i.e., the model had to be relevant to U.S. climate and swine production systems and had to be publicly available).

The Holos model considered diet (standard, low crude protein, or high-digestibility feeds) and manure handling options (anaerobic digestion, covered or uncovered slurry storage, deep pit, or solid storage). In addition, the Holos model provided an estimate of uncertainty for the model output. The MANURE model (WRI, 2009) collected the most comprehensive data and allowed for easy comparison of the impact of changes in manure handling and use on emissions of NH_3, N_2O (direct and indirect), and CH_4. On the animal side, MANURE based its calculations solely on animal numbers; feeding was not considered. The other models considered, while meeting minimum criteria, lacked any improvements over the IPCC approach. Consequently, the IPCC method was selected (i.e., Holos utilizes the IPCC Tier 1 approach for housing) with nitrogen excretion estimated using ASABE equations that account for diets.

5.3.5 Housing Emissions from Poultry Production Systems

Meat Birds

Greenhouse gas emissions within the farm boundary of a broiler chicken farm will originate almost exclusively from the animal housing, which also serves as the storage location for manure. Liu et al. (2011a) reported that for a 20-week grow-out of turkeys on litter, average daily N_2O emissions were 0.045 g (kg bodyweight)$^{-1}$, and daily CH_4 emissions were 0.08 g (kg bodyweight)$^{-1}$. Emission sources external to the housing include GHG emissions from farm vehicles. If a house is cleaned or decaked (removal of the top, crusted portion of the litter) and stored on the farm, GHG and NH_3 production and emissions could occur; Appendix 5-C provides further discussion on NH_3 emissions from housing. Practices to decake and the timing of land application of cake and litter vary from site to site and may or may not include further composting.

Laying Hens

Greenhouse gas emissions within the farm boundary of an egg farm may originate from the housing or the manure storage location. Emission sources external to the housing include GHG emissions from farm vehicles. External to the farm itself, GHG emissions result from land application of litter or stockpiling of the litter in fields prior to land application.

Laying hen housing systems without litter would likely exhibit greater emissions than litter systems, but comparison of estimates are sparse. Laying hen houses typically store excreta in a basement or may move excreta out of the house frequently (daily or more often); this would relocate emissions to a storage shed rather than change the cumulative emissions unless some form of processing (drying) took place prior to storage. Li et al. (2010) reported daily CH_4 emissions of 39.3 to 45.4 mg hen^{-1} and N_2O emissions of 58.6 mg hen^{-1} (hen bodyweight average = 1.9 kg) in a basement-type system. This compares to a litter system for a 20-week grow-out of turkeys where average daily N_2O emissions were 0.045 g kg^{-1} bodyweight and daily CH_4 emissions were 0.08 g kg^{-1} bodyweight (Liu et al., 2011a). Based on the comparison of these two studies, differences in GHG emissions from dry litter systems and wetter, stacked laying hen systems would be expected.

Management practices to reduce litter moisture offer the most promise for reducing emissions of CH_4 and N_2O. Quantitative estimates of how emissions vary with litter moisture are not available, but would likely follow similar dynamics as soil moisture content. Reuse of litter and decaking procedures might also be used as strategies to reduce emissions in the future. However, data are not available at present to use as part of a systems model.

Ammonia Emissions in Poultry Housing

As mentioned earlier, ammonia is not a greenhouse gas, however, ammonia emissions are estimated as part of the nitrogen balance approach. Meat birds are typically raised in litter systems. Litter temperature, pH, and moisture, along with the ammonium content and house ventilation rate are recognized as major factors controlling NH_3 loss from broiler litter (Elliot and Collins, 1982; Carr et al., 1990; Moore et al., 2010). There are seasonal variations in emissions, with losses tending to be greater in summer (warmer months) than in winter (Coufal et al., 2006). Bird age/size can affect litter temperature, which may influence seasonal effects on emissions (Miles et al., 2008). In addition, the formation of cake in the house can have a large impact on emissions. Miles et al. (2008) reported that extremely caked areas of the house had virtually no fluxes of NH_3. Areas of litter where anaerobic conditions develop suppress NH_3 formation and release (Carr et al., 1990). Moore et al. (2011) determined that NH_3 emissions from broiler houses averaged 37.5 g bird^{-1}, or 14.5 g kg bird marketed^{-1} (50-day old birds). The same authors estimated that of the total nitrogen output from typical broiler houses, approximately 22 percent can be associated with NH_3 emissions, 56 percent from harvested birds, and 21 percent from litter plus cake. The addition of aluminum sulfate (alum) at a rate equivalent to five to 10 percent by weight (alum/manure) reduces NH_3 emission from broiler houses by 70 percent (Moore et al., 2000) and results in heaver birds, better feed conversion, and lower mortality (Moore, 2013). Emissions of N_2O and CH_4 are dependent upon litter conditions that favor an anaerobic environment. Limited data are available documenting litter moisture content effects on N_2O and CH_4 emissions. Miles et al. (2011) demonstrated that incremental increases in litter moisture content increased NH_3 volatilization. Similarly, Cabrera and Chiang (1994) demonstrated a range in NH_3 volatilization of 32 percent to 139 percent of initial ammonium content as litter water content increased. Litter temperature is another factor that may influence GHG emissions. Miles et al. (2006) demonstrated that litter temperature affected NH_3 flux, but the study did not measure other gases. Miles et al. (2011) observed that organic bedding materials generated the least amount of NH_3 at the original moisture content when compared with the inorganic materials. The influence of bedding material at increased moisture levels was not clear across the treatments tested. But the findings suggest that choice of bedding material may also influence N_2O and/or CH_4 emissions and could potentially be used as a mitigation strategy.

5.3.5.1 Method for Estimating Emissions from Poultry Production Systems

Method for Estimating Emissions from Poultry Production Systems

Methane

- IPCC Tier 1 approach, utilizing barn capacity and manure CH_4 emissions factors per poultry type.
- IPCC emission factor for poultry enteric CH_4 production is 0. Emissions from hindgut fermentation are small and generally considered part of housing emissions.

Nitrous Oxide

- Nitrogen excretion estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O from manure in housing.

Equation 5-21: Methane Emissions from Poultry Housing (IPCC, 2006)

$$\mathbf{CH_4 = Rate \times Barn\_Capacity}$$

Where:

CH_4 = Methane emissions per year (kg CH_4 $year^{-1}$)

Rate = Manure methane emissions (kg CH_4 $head^{-1}$ $year^{-1}$)

Barn_Capacity = Capacity of barn (head)

Nitrous Oxide and Ammonia Emissions from Poultry Housing

To estimate nitrogen losses from housing, the amount of nitrogen excreted (N_{ex}) by each animal category is first estimated. Equation 5-22 and Equation 5-23 are the equations recommended by the American Society of Agricultural Engineers (ASABE) for estimating N_{ex} from broilers, turkeys, ducks, and laying hens.

Equation 5-22: ASABE Approach for Estimating Nitrogen Excretion from Broilers, Turkeys, and Ducks

$$\mathbf{N_{ex} = \sum_{x=1}^{n} \left[FI_x \times \frac{C_{CP-x}}{6.25} \right] \times (1 - N_{RF})}$$

Where:

N_{ex} = Total nitrogen excretion per finished animal (g N (finished animal)$^{-1}$)

FI_x = Feed intake per phase (g feed (finished animal)$^{-1}$)

C_{CP-X} = Concentration of crude protein of total ration in each phase (g crude protein (g (wet) feed)$^{-1}$)

N_{RF} = Retention factor for nitrogen (fraction)

Equation 5-23: ASABE Approach for Estimating Nitrogen Excretion from Laying Hens

$$\mathbf{N_{ex}} = \left(\mathbf{FI} \times \frac{\mathbf{C_{CP}}}{\mathbf{6.25}}\right) - \left(\mathbf{0.0182} \times \mathbf{Egg_{wt}} \times \mathbf{Egg_{pro}}\right)$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

FI = Feed intake (g feed (finished animal)$^{-1}$)

C_{CP} = Concentration of crude protein of total ration (g crude protein (g (wet) feed)$^{-1}$)

Egg_{wt} = Egg weight[a] (g)

Egg_{pro} = Fraction of eggs produced each day[b] (eggs hen^{-1} day^{-1})

[a] Default egg weight is 60 g for light layer strains and 63 g for heavy layer strains.
[b] Default fraction is 0.80.

The NH_3 lost from manure for meat and egg-producing birds is estimated as a fraction of N_{ex}. Koelsch and Stowell (2005) provide estimates on the typical NH_3 loss from different housing facilities as a fraction of N_{ex} (see Table 5-15). A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-15: Typical Ammonia Losses from Poultry Housing Facilities (Percent of N_{ex})

Facility Description	Applicable Species	% Loss	Facility Description	Applicable Species	% Loss
Roofed facility (litter)	Meat producing birds	25 - 50	Roofed facility (stacked manure under floor - includes storage loss)	Egg-producing birds	25 - 50

Source: Koelsh and Stowell (2005).

N_2O can also be lost from the excreted nitrogen. A quantitative method for estimating N_2O emissions from solid manure is the IPCC Tier 2 approach, which is also used for the U.S. Greenhouse Gas Inventory (Equation 5-8). This estimation method is the same as the method present in the Temporary Stack and Long-Term Stockpile and the Composting sections (see sections 5.4.1 and 5.4.2 for more details). The N_2O emission factors for poultry manure in housing is 0.001 (kg N_2O-N/ kg N) for poultry manure with or without bedding IPCC (2006).

The remaining nitrogen excreted that is not volatilized as NH_3 or lost as N_2O in housing then enters manure storage and treatment. If data are not available to track the nitrogen that is transferred along with the manure to manure storage and treatment, the nitrogen can be estimated as described in Equation 5-9. However, this equation is overestimating the nitrogen transferring to manure storage and treatment as some nitrogen will be lost in housing. This remaining total nitrogen value is an input into the N_2O equations for manure stored or treated.

5.3.5.2 Rationale for Selecting Method for Estimating Emissions from Poultry Production Systems

Miles et al. (2006) suggest that a robust model would include factors such as house management, bird size and age, cake management, pH, and litter moisture. Due to current data limitations, an NH_3 and GHG estimation model should minimally include number of animals, litter/excreta moisture content, dietary protein and fiber content, and litter/excreta pH. A variety of models applicable to poultry production facilities were identified and evaluated, including Carbon Accounting for Land Managers; CFF Carbon Calculator; CPLAN; and 4) Holos. These models were evaluated with respect to 10 criteria (see box) to determine their suitability for use in determining emissions estimates for poultry production facilities in the United States.

Model Evaluation Criteria for Poultry Production Systems

1. The model is based on well-established scientifically sound relationships between farm management inputs and emissions outputs (process-based model or mass-balance model preferable).
2. The model is relevant to U.S. climate and production systems.
3. The model can estimate CH_4, N_2O, and NH_3 emissions from poultry housing systems.
4. There is flexibility in the model to describe the production system (animals, feed, housing, and in-house manure management).
5. The model is easy to use and is designed to use easily obtainable farm information to determine emissions estimates.
6. The model includes some mitigation strategies for reducing emissions and produces realistic changes in emissions values when these changes are made within the production system.
7. There is transparency in the model calculations, and technical guidelines are available to elaborate the methodologies used to obtain the emissions estimates.
8. The model has been tested/validated with on-farm data.
9. The model works reliably (little to no errors or program crashes).
10. The model is publicly available and accessible.

Two of these criteria were considered to be critical, in that if they were not met by the model, they could not be considered for use (i.e., the model had to be relevant to U.S. climate and poultry production systems and had to be publicly available). The Holos model did consider wet or dry manure handling for laying hen operations. For all poultry types, the Carbon Accounting for Land Managers model requested information related to burning of manure and time birds spend in a free-range system. This information was then used to calculate the mass of manure available for direct and indirect emissions. No model requested information on diet or in-house litter management practices. For CH_4 emissions, only the Holos model provided an estimate of confidence of model output. Specific to estimates of poultry manure CH_4 emissions, the model had an uncertainty under 20 percent for broilers, turkeys, layers in wet manure handling systems, and layers in dry manure handling systems. Consequently, the IPCC method was selected (i.e., Holos utilizes the IPCC Tier 1 approach for housing). For N_2O emissions, the IPCC Tier 2 was used with nitrogen excretion estimated using ASABE equations that account for diets.

5.3.6 Enteric Fermentation and Housing Emissions from Other Animals

Although the majority of emissions from livestock in the United States are from cattle, sheep, swine, and poultry, emissions from other animals can also be important to consider, particularly at the entity level. Overall, populations of the animals discussed in this section (goats, American bison, llamas, alpacas, and managed wildlife) are much fewer than those of the animals discussed in prior sections. However, the availability of research on emissions from these animals allows us to explore them at least at an introductory level. At the entity level, populations of these animals may be significant enough to warrant calculating their emissions. This report recommends methods for estimating CH_4 emissions from goats and American bison (Equation 5-24 and Equation 5-25).

5.3.6.1 Goats

Enteric emissions from goat production systems were estimated by U.S. EPA (U.S. EPA, 2011) using IPCC (2006) methods to be 16 Gg CH_4 (of a total of 6,655 Gg). Emissions of manure CH_4 and N_2O from goat production were made using IPCC (2006) methods. Goats were associated with 1 Gg of manure CH_4 (of a total of 2,356 Gg) and less than 0.5 Gg of N_2O.

The impact of diet on Japanese goat enteric CH_4 emissions was measured by Bhatta et al. (2007). Goats fed a range of diets from 100 percent forage to 80 percent concentrate produced from 16.4 to 22 g CH_4 day^{-1} (5.0 to 8.2 percent of GEI).

The IPCC (2006) Tier 1 equation, presented in Equation 5-24, for estimating enteric fermentation emissions from goats is the best option for calculating emissions at the entity level.

Equation 5-24: Tier 1 Equation for Calculating Methane Emissions from Goats

$$\mathbf{CH_4 = Pop \times EF_G}$$

Where:

CH_4 = Methane emissions per day (kg CH_4 day^{-1})

Pop = Population of goats (head)

EF_G = Emission factor for goats (0.0137 kg CH_4 head^{-1} day^{-1})

5.3.6.2 American Bison, Llamas, Alpacas, and Managed Wildlife

Galbraith et al. (1998) measured enteric CH_4 from growing bison (n=5), wapiti (n=5), and white-tailed deer (n=8) fed alfalfa pellets in the winter-spring (February-March) and spring (April-May) using respiration calorimetry chambers. The bison produced an average of 86.4 g day^{-1} (6.6 percent GEI), the wapiti, 62.1 g day^{-1} (5.2 percent GEI), and the deer 23.6 g day^{-1} CH_4 (3.3 percent GEI). Using a detailed method of calculation to estimate historical bison emissions, Kelliher and Clark (2010) estimated that grazing bison would produce 72 kg CH_4 year^{-1} or 197g CH_4 day^{-1}. Hristov (2012) estimated present day bison produce 21 g CH_4 (kg DMI)$^{-1}$ day^{-1}, eat approximately 12.8 kg DM day^{-1}, and produce 268 g CH_4 day^{-1}. The differences between these estimates are differences in animal weights, DMI, limited measurements of bison emissions, and assumed CH_4 conversion factors. The U.S. EPA uses IPCC Tier 1 methodologies to estimate bison emissions, and currently Tier 1 is the best option to estimate enteric emissions.

The IPCC (2006) Tier 1 equation for estimating enteric fermentation emissions from American bison is based on the emission factor for buffalo and has been modified as recommended by IPCC to account for average weight as seen in Equation 5-25.

Equation 5-25: Tier 1 Equation for Calculating Methane Emissions from American Bison

$$\mathbf{CH_4 = Pop \times EF_{AB}}$$

Where:

CH_4 = Methane emissions per day (kg CH_4 head^{-1} day^{-1})

Pop = Population of American bison (head)

EF_{AB} = Emission factor for American bison (kg CH_4 head^{-1} day^{-1})

EF_{AB} is the IPCC emission factor for *buffalo* (0.15 kg CH_4 head^{-1} day^{-1}), adjusted for American bison based on the *ratio* of live weights of American bison (513 kg) to buffalo (300 kg) to the 0.75 power.

$$\mathbf{EF_{AB} = 55\ kg \times \left(\frac{513\ kg}{300\ kg}\right)^{0.75}}$$

The New Zealand Ministry for the Environment (2010) uses a factor of 6.4 percent of GEI to predict enteric CH_4 emissions from farmed red deer and projects an emission rate per year of 23.7 kg CH_4 head^{-1} year^{-1}. Deer are also estimated to excrete 31.0 kg N head^{-1} year^{-1} contributing toward N_2O production. The values used to make these calculations are from measurements of deer CH_4 emissions using the SF_6 tracer method. Elk, white-tailed, and mule deer enteric emissions were estimated by Hristov (2012) to be 86.4, 16, 17 g CH_4 head^{-1} day^{-1} respectively. IPCC Tier 1 is the recommended method by which these emissions should be estimated.

Adult llamas fed oat hay in a study designed to define energy requirements were found to lose 7.1 percent of GEI as enteric CH_4 (Carmean et al., 1992). Pinares-Patino et al. (2003) compared enteric CH_4 emissions measured with respiration calorimetry chambers from alpaca and sheep fed alfalfa diets and found the alpaca produced 14.9 g CH_4 day^{-1} (5.1 percent of GEI) and the sheep produced 18.8 g CH_4 day^{-1} (4.7 percent of GEI). When grazing a perennial ryegrass/white clover pasture, the alpaca produced 22.6g CH_4 day^{-1} (9.4 percent GEI) compared to 31.1 g CH_4 day^{-1} (7.5 percent GEI) for sheep. The authors attribute the high conversion of GEI to CH_4 from the alpaca to grazing selectivity on pasture; the alpaca were observed to select "more structural plant parts."

5.3.7 Factors Affecting Enteric Fermentation Emissions

A number of factors may influence enteric fermentation and resulting CH_4 emissions. A thorough review of such factors is outside the scope of this document, but key factors have been reviewed by others (Monteny et al.; (2006), Beauchemin et al.; (2008), Eckard et al.; (2010), and Martin et al.; (2010)) and are discussed briefly below.

Benchaar et al. (2001) used the rumen digestion model of Dijkstra et al. (1992), as modified by Benchaar et al. (1998), and the CH_4 prediction system of Baldwin (1995) to estimate the effects of dietary modifications on the enteric CH_4 production of a 500 kg dairy cow. The model predicted enteric CH_4 production based on a ruminal H balance. Inputs into the model included the following: daily DMI; chemical composition of the diet; solubility and degradability of protein and starch in the diet; degradation rates of protein, starch, and NDF; ruminal volume; and fractional passage rates of solids and liquid fractions from the rumen. Values modified in the simulations were DMI, dietary forage, concentrate ratio, starch availability (barley vs. corn), stage of maturity of forage, form of forage (hay or silage), particle size of alfalfa, and ammonization of cereal straw. The modeled effects of dietary changes on enteric CH_4 emissions in diets fed to dairy cows are presented in Table 5-16.

There are many factors that affect enteric CH_4 emissions but the most critical factors are the level of dry matter intake, the composition of the diet, and the digestibility of the dry matter, as illustrated in Table 5-16.

Table 5-16: Summary of Effects of Various Dietary Strategies on Enteric CH_4 Production in Dairy Cows using Modeled Simulations

Strategy	CH_4 Variation (per unit of GEI)	CH_4 Variation (per unit of DE)
Increasing DMI	-9 to -23%	-7 to -17%
Increasing concentrate proportion in the diet	-31%	-40%
Switching from fibrous concentrate to starchy concentrate	-24%	-22%
Increased forage maturity	+15%	-15%
Alfalfa vs. timothy hay	+28%	-21%
Method of forage preservation (ensiled vs dried)	-32%	-28%
Increased forage processing (smaller particle size)	-21%	-13%
Ammoniated treatment of poor quality forage(straw)[a]	x 5	x 2
Protein supplementation of poor quality forage (straw)	× 3	× 1.5

Source: Benchaar et al., (2001), Table 12.
[a] Effects are due to significant increase in hay digestibility with no change in DM intake.

Dietary Fat: Many studies have demonstrated that supplemental fat can decrease enteric CH_4 emissions in ruminants. In a review of studies, Beauchemin et al. (2008) noted that enteric CH_4 emissions (g $[kg\ DMI]^{-1}$) decreased by approximately 5.6 percent for each one percent increase in fat added to the diet. In a larger review, Martin et al. (2010) reported a decrease of 3.8 percent (g $[kg\ DMI]^{-1}$) with each one percent addition of fat. Lovett et al. (2003) reported that total daily emissions decreased from 0.19 to 0.12 kg CH_4 $head^{-1}$ (reported as 260 to 172 L CH_4 $head^{-1}$) (6.6 and 4.8 percent of GEI) from steers fed diets containing 0 or 350 g of coconut oil, respectively. This effect was consistent regardless of dietary forage concentration (65, 40, and 10 percent of DM).

Although added fat may reduce enteric CH_4 emissions, ruminants have a low tolerance for dietary fat. Thus, total fat level in the diet must be maintained below eight percent of dietary DM. Some sources of fat appear to have some protection against biohydrogenation by ruminal microbes and thus may be better tolerated (Corrigan et al., 2009; Vander Pol et al., 2009).

Grain Source, Grain Processing, Starch Availability: Grain source and grain processing method can also affect enteric CH_4 losses. In general, the greater the ruminal starch digestibility, the lower the enteric CH_4 emissions. At constant energy intake (2 x maintenance), Hales et al. (2012) reported approximately 20 percent lower (2.5 vs. 3.0 percent of GEI) enteric CH_4 emission in cattle fed typical high-concentrate (75 percent corn) steam flaked corn (SFC) based finishing diets than in steers fed dry-rolled corn (DRC) based finishing diets. Based on the rumen stoichiometry of Wolin (1960), Zinn and Barajas (1997) estimated that CH_4 production per unit of glucose equivalent fermented in the rumen also decreased with more intensive grain processing (i.e., coarse, medium, or fine flakes). Similar responses were noted with the feeding of high-moisture corn compared with DRC (Archibeque et al., 2006). Somewhat in contrast, Beauchemin and McGinn (2005) reported lower enteric CH_4 emissions from feedlot cattle fed DRC-based diets (2.81 percent of GEI) than from cattle fed steam-rolled barley-based diets (4.03 percent of GEI), possibly the result of lower ruminal pH on the corn-based diet (5.7 vs. 6.2, respectively; (Van Kessel and Russell, 1996) and/or higher NDF in the barley diet. Enteric CH_4 emissions were 38 percent (barley) to 65 percent (corn) lower in high-concentrate (nine percent silage) finishing diets than on grower (70 percent silage) diets.

Feeding Coproduct Ingredients: Distillers grains with solubles (DGS) and other coproducts of the milling and ethanol industries are widely used as cattle feeds. The effects of feeding 30 to 35 percent DGS (DM basis) on enteric CH_4 emission have been variable, ranging from a significant decrease of 25 to 30 percent (McGinn et al., 2009) to no effect (Hales et al., 2012), to an increase (Hales et al., 2013). These differing results were probably due to differences in forage and fat intake. In the study by McGinn et al. (2009) the diet contained 65 percent silage, and dietary fat intake increased by approximately three percentage units[8] when dried DGS were added to the diet. In contrast, Hales et al. (2012; 2013) fed diets that contained only 10 percent forage and were equal in total fat concentration.

Roughage Concentration and Form: The concentration and form of roughage in the diet will affect both enteric and manure CH_4 production (Hales et al., 2014). Using a ruminal volatile fatty acids (VFA) stoichiometry model, Dijkstra et al. (2007) suggested that CH_4 losses from carbohydrates substrates (g kg^{-1} substrate) in a concentrate diet with ruminal pH variation and a pH of 6.5 were 2.11, 3.18, 3.38, and 3.10 for starch, soluble sugars, hemicellulose, and cellulose, respectively. Similarly, with dairy cows, Moe and Tyrrell (1979) reported that enteric CH_4 production per unit carbohydrate digested was three times greater for cellulose than for hemicellulose. Aguerre et al. (2011) found that lactating dairy cattle emitted more CH_4 when the forage:concentrate ratio was changed from 47:53 to 68:32, 0.54 kg CH_4 day^{-1} vs. 0.65 kg CH_4 day^{-1} respectively.

In general, as the concentration of forage in the diet increases, enteric CH_4 production increases and the quantity of volatile solids excreted increases. Using a micrometeorological mass difference method, Harper et al. (1999) reported CH_4 emissions of 230 g $animal^{-1}$ daily (7.7 to 8.1 percent of GEI) in feeder cattle on pasture, but only 70 g $head^{-1}$ daily (1.9 to 2.2 percent of GEI) in cattle fed high-concentrate diets. Measured CH_4 losses for pasture cattle were higher than values predicted using the IPCC (1997; 2006) CH_4 conversion factors (MCF or Ym), or Australian methodology (NGGIC, 1996). In contrast, measured CH_4 losses for feedlot cattle were about 67 percent of those estimated using the IPCC (2006) Ym of three percent of GEI or the Australian methodology (NGGIC, 1996), but were similar to values reported by Branine and Johnson (1990), Blaxter and Wainman (1964), and Hales et al. (2012; 2013; Hales et al., 2014).

Enteric fermentation of tropical grasses and legumes may also be different than predicted by IPCC or national GHG inventory methods. Kennedy and Charmley (2012) measured enteric CH_4 production of cattle fed Australian tropical grasses and legumes to be 5.0 to 7.2 percent of GE intake which is similar to IPCC (2006) Tier 2 estimates (5.5 to 7.5 percent of GE intake) of cattle fed forage diets but somewhat lower than the Australian National Greenhouse Accounts National Inventory Report (2007) of 8.7 to 9.6 percent of GE intake.

Blaxter and Wainman (1964) compared the effects of feeding diets with six varying hay: flaked corn ratios (100:0, 80:20, 60:40, 40:60, 20:80, 5:95) on enteric CH_4 emissions when fed at two times the maintenance level of intake. CH_4 emissions as a percentage of GEI increased slightly between the 100:0 diet (7.44 percent) and the 60:40 diet (8.17 percent), then decreased to the 5:95 diet (3.4 percent).

In Ireland, Lovett et al. (2003) reported total daily enteric CH_4 emissions of 0.15, 0.19, and 0.12 kg $head^{-1}$ (reported as 207, 270, and 170 L $head^{-1}$) for heifers fed diets containing 65, 40, and 10 percent forage (the remainder as concentrate), respectively. As a percentage of GEI, losses were 6.1, 6.6, and 4.4 percent, respectively.

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[8] The term "percentage units" in this document is used to refer to changes in diets or emissions that are *not* proportional to their baselines. For example, a reduction in emissions from three percent to one percent is a 2 "percentage unit" reduction or a *67 percent reduction*.

Using steers fed all-forage diets, Ominski et al. (2006) reported that, within the range of forage qualities tested (alfalfa-grass silage containing 61, 53, 51, or 46 percent NDF, DM basis), enteric CH_4 emissions of steers, as a percentage of GEI, were not significantly affected by NDF content (5.1 to 5.9 percent), although daily CH_4 production tended to be highest for the 53 percent NDF diet (0.12, 0.15, 0.13, and 0.14 kg head^{-1} day^{-1}, respectively). Similarly, using grazing sheep, Milano and Clark (2008) reported no effect of forage quality (rye grass – 52 or 47 percent NDF, 77 or 67 percent organic matter [OM] digestibility) on enteric CH_4 emissions.

Although dietary forage quality may sometimes not affect enteric CH_4 emissions, it will affect forage digestibility and thus fecal excretion of volatile solids. Thus, feeding more digestible forages or concentrates may decrease GHG emissions from manure.

Level of Intake: Blaxter and Wainman (1964) compared the effects of feeding six diets at two levels of intake. Enteric CH_4 emissions, as a percent of GEI, were 23 percent greater in steers fed at maintenance than in steers fed at 2X maintenance (8.1 vs. 6.6 percent of GEI, respectively). However, in a study evaluating emissions from cattle fed ryegrass diets, Milano and Clark (2008) reported that as DMI increased from 0.75 percent of maintenance to 2X maintenance, enteric CH_4 emissions (g day^{-1}) increased linearly (r^2 = 0.80 to 0.84). Emissions as a percentage of GEI were not affected by DMI, and ranged from 4.9 to 9.5 percent of GEI (15.9 to 30.4 g [kg DMI]$^{-1}$).

Using a high-forage (70 percent barley silage) or medium-forage (30 percent silage) diet fed at levels from 1X to approximately 1.8X maintenance, Beauchemin and McGinn (2006b) noted that enteric CH_4 emissions, as a percent of GEI, decreased by approximately 0.77 percentage units for each unit increase in feed intake (expressed as level of feed intake above maintenance). This was less than the estimate using the Blaxter and Clapperton (1965) equation (0.93 to 1.28 percent percentage units) or the 1.6 percentage units suggested by Johnson and Johnson (1995).

Feed Additives and Growth Promoters: Cooprider et al. (2011) noted that the daily CH_4 and manure N_2O production of cattle fed through a “natural” program with no use of antibiotics, ionophores, or growth promoters were similar to cattle fed in more traditional systems that used anabolic implants and diets that contained ionophores and beta-agonists. However, typical cattle had greater average daily weight gain (1.85 vs. 1.35 kg day^{-1}) and thus took 42 fewer days to reach the same end point (596 kg body weight [BW]). Thus, overall, cattle fed using modern growth technologies had 31 percent lower GHG emissions per head. CH_4 emissions kg of BW gain^{-1} was 1.1 kg greater for the “natural” cattle (5.02 vs. 3.92 CO_2-eq kg BW gain^{-1}) than the traditional cattle.

Monensin decreases enteric CH_4 emissions in finishing cattle by 10 to 25 percent (Tedeschi et al., 2003; McGinn et al., 2004). However, in feedlot cattle the effects appear to be transitory, lasting for 30 days or less (Guan et al., 2006). In contrast, Odongo et al. (2007) reported that monensin (24 ppm) in dairy diets decreased enteric CH_4 by seven to nine percent for up to six months. Waghorn et al. (2008) found no effect of monensin controlled-release capsules on CH_4 production of pasture-fed dairy cows, and Hamilton et al. (2010) also found no change in enteric CH_4 production from monensin fed to dairy cows offered a total mixed ration.

A number of studies have demonstrated that a variety of halogenated analogues have the potential to dramatically decrease ruminal CH_4 production (Johnson, 1972; Trei et al., 1972; Johnson, 1974; Cole and McCroskey, 1975; Tomkins and Hunter, 2004; Tomkins et al., 2009). In general the effect was greater in cattle fed high-forage diets than in cattle fed high-concentrate diets. When CH_4 losses were dramatically reduced, a significant quantity of hydrogen could be lost (one to two percent of GEI) via eructation, suggesting an alternative electron sink is also needed. In general, the compounds did not improve production efficiency significantly. In addition, the potential toxicity of these compounds made them impractical for routine use.

A number of nitrocompounds (nitropropanol, nitroethane, nitroethanol) have also significantly decreased ruminal CH_4 production in vitro (Anderson et al., 2003), with a concomitant increase in hydrogen production/release. The effect appeared to be enhanced when a nitrate reducing bacterium was added to the culture (Anderson and Rasmussen, 1998).

Several studies have suggested that feeding of condensed tannins can decrease enteric CH_4 production by 13 to 16 percent; either through a direct toxic effect on ruminal methanogens or indirectly via a decrease in feed intake and diet digestibility (Eckard et al., 2010). Tannins may also shift nitrogen excretion away from urine to feces and inhibit urease activity in feces, which could potentially decrease NH_3 and N_2O emissions from manure (Powell et al., 2009; Powell et al., 2011).

Feeding yeast cultures, enzymes, dicarboxylic acids (fumarate, malate, acrylate), and plant secondary compounds, such as saponins, may decrease enteric CH_4 emissions under some feeding conditions (McGinn et al., 2004; Beauchemin and McGinn, 2006a; Ungerfeld et al., 2007; Beauchemin et al., 2008; Eckard et al., 2010; Martin et al., 2010).

Novel Microorganisms and their Products: Klieve and Hegarty (1999) noted that enteric CH_4 production may be biocontrolled directly by use of viruses and bacteriocins. Lee et al. (2002) reported that a bacteriocin (Bovicin HC5) from *Streptococcus bovis* reduced in vitro CH_4 production by up to 50 percent. It appeared, that in contrast to results with monensin, the ruminal microorganisms did not adapt to the bacteriocin.

Australian researchers have suggested that vaccinating against methanogens can decrease CH_4 emissions. However, the results have not been consistent (Wright et al., 2004; Eckard et al., 2010) because efficacy is dependent on the specific methanogen population and that is dependent on diet, location, and other factors.

Genetics: As previously noted, several studies have suggested that cattle selected for lower RFI (i.e., increased feed use efficiency) tend to have lower ruminal enteric CH_4 production (Nkrumah et al., 2006; Hegarty et al., 2007), although the effect may depend on stage of production (lactation vs. dry and pregnant) and/or quality of the diet consumed (Jones et al., 2011). RFI is moderately heritable (0.28 to 0.58) (Moore et al., 2009), thus it might be possible to genetically select for animals with lower enteric CH_4 production. However, Freetly and Brown-Brandl (2013) found higher CH_4 emissions from more efficient animals. Thus, more information is needed to define under what conditions CH_4 emissions are related to feed efficiency or to genetics.

Factors Affecting Emissions from Sheep

Sheep, like cattle, are ruminant animals and thus the same dietary factors will positively or negatively affect emissions from enteric fermentation.

Factors Affecting Emissions from Swine

Dietary modifications can effectively reduce nitrogen excretions and mitigate air emissions (especially NH_3, a precursor for N_2O) from livestock operations (Sutton et al., 1996; Canh et al., 1998b). Feeding strategies to reduce nitrogen excretions include reduced CP diets supplemented with synthetic amino acids (AA) (Panetta et al., 2006), and modifying the dietary electrolytes to reduce urinary pH (Canh et al., 1998a). In both hog *and* poultry operations, reductions in NH_3 emissions have been reported by supplementing with AA and reducing CP in diets.

Reducing dietary CP content has been shown to be an effective way to reduce the amount of nitrogen excreted (Lenis, 1993; Hartung and Phillips, 1994). This can be achieved without any negative effect on animal performance by supplementing with an improved synthetic AA balance, resulting in a reduction of excess CP excreted (Canh et al., 1998b; Ferket et al., 2002). In U.S.-type diets (corn-soybean meal based) the most limiting amino acids are Lysine, Methionine, Threonine, and Tryptophan, followed by Isoleucine, Valine, and Histidine (Outor-Monteiro et al., 2010). Sutton et al. (1996) reported that nitrogen excretion decreased by 28 percent when diet CP content decreased from 13 percent to 10 percent (corn-soybean meal) for growing-finishing pig diets supplemented with Lys, Met, Thr, and Trp. Several studies reported reductions in nitrogen excretion and subsequent decreases in NH_3 emissions in non-ruminants (swine and poultry) (Hartung and Phillips, 1994; Canh et al., 1997; Canh et al., 1998a; Canh et al., 1998b; Hayes et al., 2004). Powers et al. (2007) observed that, as a result of feeding reduced CP diets with increased amounts of synthetic AA, NH_3 emissions were reduced by 22 percent (three AA) and 48 percent (five AA) compared with the control diet containing only one AA, and diet had no effect on pig performance.

Canh et al. (1998b) and Ndegwa et al. (2008) reported that some nitrogen excretion could be shifted from urine to feces by increasing dietary fiber content, or by reducing dietary nitrogen content, with no significant differences in animal performance or growth. Urinary nitrogen is predominantly inorganic in nature and fecal nitrogen is mostly organic. The conversion of urea from urine to NH_3 is a fast process, while conversion of organic nitrogen to volatile NH_3 in feces is a slow process.

The reduction in NH_3 emission associated with lower CP diets not only comes from reduction in nitrogen excretion, but also from lower manure pH. Portejoie et al. (2004) reported that slurry pH decreased by 1.3 units when dietary CP decreased from 20 to 12 percent, and slurry from pigs fed the lower CP diet had a higher DM content and lower TAN and TKN contents. Le et al. (2008), Hanni et al. (2007), and Canh et al. (1998b) also reported that lower manure pH resulted from feeding lower CP in diets. It should be noted that water intake was often restricted in earlier studies.

Aarnink and Verstegen (2007) summarized four dietary strategies to reduce NH_3 emissions: 1) lowering CP intake in combination with the addition of limiting AA; 2) shifting nitrogen excretion from urine to feces by including fermentable carbohydrates in the diet; 3) lowering urinary pH with the addition of acidifying salts to the diet; and 4) lowering feces pH with the inclusion of fermentable carbohydrates in the diet. They claimed that by combining these strategies, NH_3 emissions in growing-finishing pigs could be reduced by a total of 70 percent. To reduce odor from pig manure, Le et al. (2007) suggest that dietary sulfur-containing AA should be minimized to just meet the recommended requirements.

Current research has concentrated on farm production efficiency and reducing NH_3 emissions; little has focused on GHG emissions mitigation (Bhatti et al., 2005). Ball and Möhn (2003) showed that low CP diets can reduce total GHG emissions from growing pigs by 25 to 30 percent (directly from the animals as well as from the manure after excretion) and from sows by 10 to 15 percent. Atakora et al. (2003) reported a 27.3 percent decrease in CH_4 emissions in pigs fed 16 percent CP (supplemented with AA) diets, compared with 19.0 percent CP diets. Atakora et al. (2004) reported that the CO_2 equivalents emitted by finishing pigs and sows fed wheat-barley-canola diets were reduced by 14.3 to 16.5 percent when feeding the reduced CP, AA-supplemented diets, and were similar for finishing pigs and sows. The reduction was only 7.5 percent when feeding the corn-soybean meal-based reduced CP diet. Misselbrook et al. (1998) found that CH_4 emissions during storage were less at low than at a high dietary CP content. The emission of CH_4 was significantly related to content of dry matter, total carbon, and VFA in the manure. Misselbrook et al. (1998) claimed that the 50 percent reduction in CH_4 emission from the slurry observed when pigs were fed the lower CP diet was probably the result of the reduced volatile fatty acids (VFA) content of the slurry, and CH_4 emissions were more closely related to VFA content than to total carbon content. There appears to be a close relationship between fermentable carbohydrates in the diet and CH_4 production (Kirchgessner et al., 1991). Manure pH also influences CH_4 production. Kim et al. (2004) noted a 14 percent reduction in CH_4 emission when ideal pH was reduced one unit through addition of acidogenic calcium and phosphorus sources to pig diets. Increasing fermentable carbohydrate levels in the diet to lower the pH of manure, with the goal of reducing NH_3 emissions, might increase CH_4 production (Aarnink and Verstegen, 2007). Canh et al. (1998a) observed that for each 100-g increase in the intake of dietary non-starch polysaccharide (NSP), the slurry pH decreased by approximately 0.12 units and the NH_3 emission from slurry decreased by 5.4 percent when dietary NSP ranged from 150 to 340 g NSP kg DM^{-1}.

Feeding of dried distillers grains with solubles (DDGS) has become common practice in the swine industry. Li et al. (2011) demonstrated that feeding diets containing 20 percent DDGS increased emissions of CH_4 but not N_2O when compared to control diets without DDGS. Observed increases in CH_4 emissions approximated 18 percent. Ammonia emissions resulting from feeding 20 percent DDGS were either higher or lower than diets without DDGS, depending on the form of trace minerals included in the diet. Diets including inorganic forms of trace minerals had seven percent greater NH_3 emissions, while feeding organic forms of trace minerals decreased NH_3 emissions almost 20 percent compared to control diets (Liu et al., 2011a).

In a recent meta-analysis, Liu et al. (2011a) used 32 data points in a subgroup of studies that included diet CP information to analyze the effect of diet CP on GHG emissions. Three factors (diet CP, geographic region, and swine production phase) were considered in the regression analysis. Diet CP was not a significant factor. Emissions of CH_4 are positively correlated with diet crude protein in swine production, most significantly for lagoon and slurry storage systems (Liu et al., 2011a). Clark et al. (2005) determined that reducing dietary CP may actually increase CH_4 emissions, so results are varied. It had been expected that a lower CP diet may result in lower nitrogen excretion, and thus might be able to reduce N_2O emissions from manure. However, this hypothesis was not supported by the results of the meta-analysis.

Diet formulation at each stage of the life cycle influences nutrients excreted in manure, as well as emissions that result from that manure during storage and potentially following land application. From a modeling perspective, the focus needs to be on management factors, including diet formulation and manure handling practices.

Feed efficiency improvements can reduce emissions throughout the entire food production cycle by reducing the amount of feed needed for meat production, thereby reducing inputs into feed production as well as reducing manure nutrients that must be managed. Feed efficiency is the product of genetics and environment (management). Genetic differences are difficult to assess, because this information is retained by companies. Genetic improvements are not insignificant over time and may in fact be a larger contributor to gains than management. However, from a modeling perspective, the focus needs to be on management factors, including diet formulation and in-house manure/litter practices. Feed efficiency could be a model component in the future once more data on the impacts of feed efficiency on GHG emissions are available.

Factors Affecting Emissions from Meat Birds

Emissions of both N_2O and NH_3 can be restricted by reducing the litter nitrogen content through diet modification. Ferguson et al. (1998a; 1998b) fed reduced dietary protein diets to broiler chickens. Although performance was hindered in both studies, NH_3 concentration and litter nitrogen content were reduced significantly as a result of the low-protein diets. Applegate et al. (2008) reported similar litter nitrogen effects when turkey toms were fed reduced-protein diets. No performance differences were observed. These diets were then fed to turkey toms by Liu et al. (2011a), who observed a 12 percent reduction in NH_3 emissions as a result of reducing cumulative nitrogen intake by 9 percent. Feeding specific AA allowed for similar nitrogen intakes across treatments, but reduced NH_3 emissions by 25 percent (Liu et al., 2011a) and nitrogen in litter by 12 percent (Liu et al., 2011b), because nitrogen was better utilized by the birds. Across all diets, N_2O

emissions made up less than one percent of nitrogen output (Liu et al., 2011b), suggesting that reducing dietary nitrogen may have less influence on N_2O emissions than other factors.

Factors Affecting Emissions from Laying Hens

Diet factors can alter air emissions from laying hen facilities. Much of the work to date has focused on reducing NH_3 emissions. Roberts et al. (2007) showed that inclusion of dietary corn DDGS, wheat middlings, or soy hulls lowered the seven-day cumulative manure NH_3 emission from 3.9 g kg of dry manure^{-1} for the control to 1.9, 2.1, and 2.3 g kg of dry manure^{-1}, respectively; it also lowered the daily NH_3 emission rate. Reducing the CP content by one percent had no measurable effect on NH_3 emission. Wu-Haan et al. (2007b) fed a reduced-emissions diet containing 6.9 percent of a $CaSO_4$-zeolite mixture and slightly reduced protein to 21-, 38-, and 59-week-old Hy-Line W-36 hens; they observed that daily NH_3 emissions from hens fed the reduced-emissions diets (185.5, 312.2, and 333.5 mg bird^{-1}) were less than emissions from hens fed the control diet (255.1, 560.6, and 616.3 mg bird^{-1}) for trials 1, 2, and 3, respectively. Total nitrogen excretion from hens fed the control and reduced-protein diets was not different (Wu-Haan et al., 2007a). Because of the acidifying nature of the diets, the mass of nitrogen remaining in excreta following a three-week storage period was less from hens fed the control diet than from hens fed the reduced-protein diet (Wu-Haan et al., 2007a). Li et al. (2010) found that feeding corn DDGS decreased the mass of NH_3 emitted daily by 80 mg hen^{-1}(592 vs. 512 mg hen^{-1} day^{-1} for zero percent and 20 percent DDGS, respectively), and by 14 percent per egg produced, and daily CH_4 emissions by 13 to 15 percent (39.3 vs. 45.4 mg hen^{-1} day^{-1}; and 0.70 vs. 0.82 mg g egg^{-1} day^{-1}).

5.3.8 Limitations and Uncertainty in Enteric Fermentation and Housing Emissions Estimates

At the entity level, uncertainty in enteric CH_4 production in cattle typically results from, lack of precision in estimating energy intake, feed type and intake, characteristics of particular feedstuffs (i.e., acid detergent fiber, starch, etc.), DE, maximum possible CH_4 emissions, CH_4 conversion factors (Ym), synergies or countereffects between mitigation options, and net energy expenditure by the animal. The assumptions about implications of dietary changes on enteric CH_4 production are based on literature values (including empirical field studies) and may not be indicative of true changes in emissions for particular animal types, as this will vary depending on an individual animal's health, management practices, animal activities, and baseline diet. For swine, goats, American Bison, llamas, alpacas, and managed wildlife, the recommended estimation methods for emissions from enteric fermentation are based on the IPCC Tier 1 approach, which has an uncertainty of 30 to 50 percent.

Methane emissions from dairy cattle housing areas are estimated using equations from DairyGEM (IFSM). In predicting emissions, uncertainty will result from a lack of precision in estimating excreted volatile solids and nitrogen excreted, pH, temperature, air velocity, and surface area of exposed manure, bedding pack, CH_4 conversion factors (MCFs), and maximum CH_4-producing capacity for manures. Comparison of modeled values with on-farm evaluations has found the model predicts on-farm emissions within five to 20 percent (unpublished data).

Methane emissions from poultry housing areas are estimated using the IPCC Tier 1 method. Uncertainty in predictions of emissions result from a lack of precision in estimating feed intake, nitrogen excreted and volatile solids, MCF, volatilization fraction, and in some instances emission factors that were chosen in the model. Unfortunately there is a lack of published information related to GHG emissions from poultry and to the best of our knowledge this model has not been validated/tested using on-farm data.

Much of the published uncertainty information in inventory guidance, such as IPCC Good Practice Guidance (IPCC, 2000) and in the U.S. National GHG Inventory (U.S. EPA, 2013), focus on uncertainties present in calculating inventories at the regional or national scale, many of which do not translate to the entity level. Some of the sources of uncertainty at the regional or national scale included variability in native vegetation eaten by grazing animals, assumptions about the types of feed farmers provide for animals (including the practice of including nutritional supplements), management practices such as housing options and daily animal activity, average animal weights, and animal populations. The quantity of uncertainty at larger scales is difficult to define, dependent on both the accuracy in reporting practices and experts' understanding of the implications of management practices and the accuracy of particular estimation methodologies. Consistent improvement in reporting practices can help remove some of this uncertainty.

Available default values and uncertainty information is included in Table 5-17.

Table 5-17: Available Uncertainty Data for Emissions from Housing and Enteric Fermentation

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Daily Milk Production	Milk	kg milk/animal/day		3%	5%			Expert Assessment
Supplemental Fat (feedlot)	S.Fat	Percent				2	4	Expert Assessment
Maximum daily emissions for dairy cows	E_{max}	MJ/head	45.98					Mills et al. (2003)
Typical Ammonia Losses from Dairy Housing Facilities –Open dirt lots (cool, humid region)	NH_3 loss	Percent of N_{ex}				15%	30%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Open dirt lots (hot, arid region)	NH_3 loss	Percent of N_{ex}				30%	45%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	NH_3 loss	Percent of N_{ex}				5%	15%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (shallow pit under floor)	NH_3 loss	Percent of N_{ex}				10%	20%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (bedded pack)	NH_3 loss	Percent of N_{ex}				20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	NH_3 loss	Percent of N_{ex}				30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities – Open dirt lots (cool, humid region)	NH_3 loss	Percent of N_{ex}				30%	45%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities – Open dirt lots (hot, arid region)	NH_3 loss	Percent of N_{ex}				40%	60%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities –Roofed facility (bedded pack)	NH_3 loss	Percent of N_{ex}				20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	NH_3 loss	Percent of N_{ex}				30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	$\%NH_3$ loss	Percent of N_{ex}				5%	15%	Koelsh and Stowell (2005)

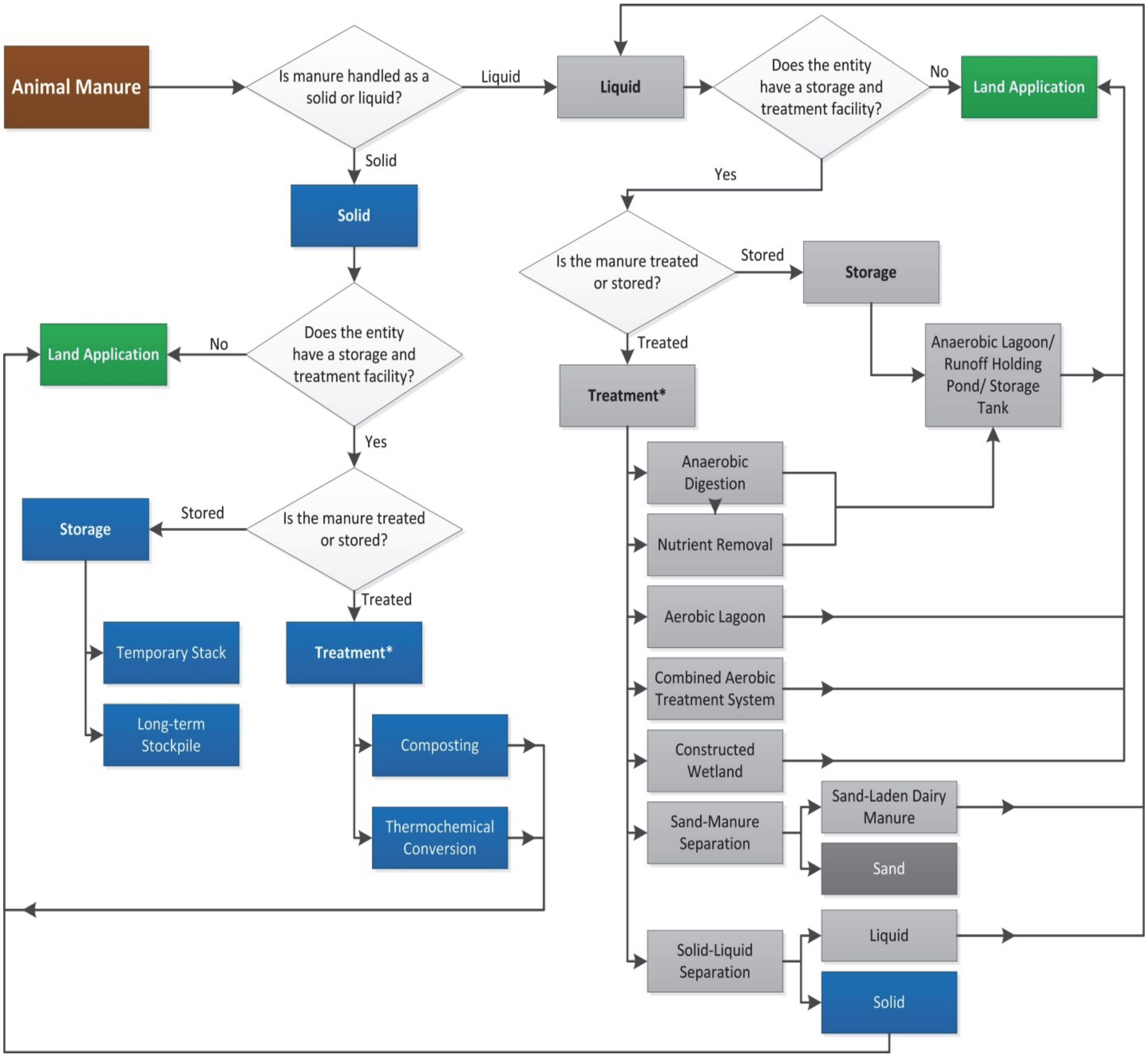
Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (shallow pit under floor)	$\%NH_3$ loss	Percent of N_{ex}				10%	20%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (bedded pack)	$\%NH_3$ loss	Percent of N_{ex}				20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	$\%NH_3$ loss	Percent of N_{ex}				30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Poultry Housing – Roofed facility (litter) (Meat Producing birds)	$\%NH_3$ loss	Percent of N_{ex}				25%	50%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Poultry Housing – Roofed facility (stacked manure under floor, includes storage loss) (Egg-producing birds)	$\%NH_3$ loss	Percent of N_{ex}				25%	50%	Koelsh and Stowell (2005)
Methane Emissions from Goats – Emission factor for goats	EF_G	kg CH_4/head/day	0.0137					IPCC (2006)

5.4 Manure Management

Use of manure as a source of plant nutrients reduces the need for purchased commercial fertilizer. Manure storage allows for manure applications to land to be synchronized with crop cultural needs. This practice reduces the potential for soil compaction due to poor timing of manure application (wet soil conditions) and makes more efficient use of farm labor. Many animal manure storage or treatment structures create anaerobic conditions that result in the production and release of GHGs and odors. Manure that is recycled to the land base can have potential negative effects on water quality (both surface and ground water).

Manure storage and treatment, as a component of manure management systems, plays a critical role in GHG emissions. At the entity level, various manure storage and treatment approaches will lead to different amounts of GHG emission. Animal manure can be classified into two categories based on their physical properties: *solid*, defined as dry matter above 15 percent; and *liquid*, defined as dry matter of less than 15 percent (including liquid manure with a dry matter of less than 10 percent and *slurry* manure with a dry matter between 10 and 15 percent). Three solid manure storage/treatment practices (temporary stack/long-term stockpile, composting, and thermo-chemical conversion) and eight liquid manure storage/treatment practices (aerobic lagoon, anaerobic lagoon/runoff holding pond/storage tanks, anaerobic digestion, combined aerobic treatment system, sand-manure separation, nutrient removal, solid-liquid separation, and constructed wetland) were evaluated and the emission estimation methods are presented. At the farm entity level, several practices are often strategically combined to treat manure. In order to provide tools to evaluate these scenarios, activity data (i.e., mass flow data and chemical and physical characteristics of influent and effluent, environmental temperature, pH, and total nitrogen) from individual practices will be used to link practices in the combined system for individual farm entities. A schematic structure of possible combinations of manure storage and treatment practices at the entity level is presented in Figure 5-7. As illustrated in the figure, manure can be handled as a solid or liquid. For each stream, the manure can be applied directly to land, stored, or treated before storage or land application. In some practices, solids are separated from the liquid manure stream and treated using a solids handling system.

Figure 5-7: Schematic Structure of Possible Combination of Manure Storage and Treatment Practices



Note: Individual practices could be combined together to treat manure based on the need at the entity level.

Each manure management practice is described as an individual unit practice in this document. The references for estimation of GHG emission for individual practice are listed in Table 5-18.

Table 5-18: List of Individual Manure Storage and Treatment Practices

Section	Storage and Treatment Practices	Major References for GHG Estimation
Solid manure		
5.4.1	Temporary and long-term storage	IPCC (2006); U.S. EPA (2011)
0	Composting	IPCC (2006); U.S. EPA (2011)
Liquid manure		
5.4.3	Aerobic lagoon	IPCC (2006); U.S. EPA (2011)
5.4.4	Anaerobic lagoon/runoff holding ponds/storage tanks	Sommer et al. (2004)
5.4.5	Anaerobic digestion with biogas utilization	IPCC (2006); CDM (2012)

Section	Storage and Treatment Practices	Major References for GHG Estimation
5.4.6	Combined aerobic treatment system	Vanotti et al. (2008)
5.4.7	Sand–manure separation	
5.4.8	Nutrient removal	
5.4.9	Solid–liquid separation	Ford and Fleming (2002)
5.4.10	Constructed wetland	Stein et al. (2006; 2007b) Stone et al. (2002; 2004)
5.4.11	Thermo-chemical conversion	

The remainder of this section presents the method for estimating GHGs from the sources listed in Table 5-18. For each source of GHGs with an estimation method, the following information is provided:

- **Overview of the GHG Source and the Resulting GHGs.** This section provides an overview of manure management technology, the resulting GHG emissions, and the methodology proposed for estimating the emissions.
- **Rationale for Selected Method**. This section presents the reasoning for the selection of the method recommended in this report.
- **Activity Data**. This section lists the activity data required for estimating GHGs at the entity level.
- **Ancillary Data.** This section lists ancillary data such as CH_4 conversion factors (MCF) and maximum CH_4 production capacity (B_0).
- **Method**. This section provides detailed descriptions, including equations for the selected methods.
- For each source of GHGs without an estimation method, a qualitative overview is provided. Methods for estimating NH_3 emissions are provided in Appendix 5-C.

5.4.1 Temporary Stack and Long-Term Stockpile

5.4.1.1 Overview of Temporary Stack and Long-Term Stockpiles

Method for Estimating Emissions from Manure Storage and Treatment – Temporary Stack and Long-Term Stockpile

Methane

- IPCC Tier 2 approach using IPCC and U.S. EPA Inventory emission factors, utilizing monthly data on volatile solids and dry manure. Volatile solids content can be obtained from sampling and lab testing.
- Method is only readily available method.

Nitrous Oxide

- IPCC Tier 2 approach using U.S.-based emission factors and monthly data on volatile solids, total nitrogen, and dry manure.
- No specific models exist; method is the only readily available method.

Management methods for stored manure are differentiated by the length of time they are stockpiled (i.e., temporary stack and long-term storage). Temporary stack is a short-term manure storage method that is used to temporarily hold solid manure when bad weather prohibits land application, and/or when there is limited availability of cropland for manure application. With temporary stack, the manure is removed and applied to land within a few weeks of piling. Temporary storage is not a preferred method to store manure because it requires the manure to be handled twice.

Long-term storage is a permanent manure storage method in which solid manure is piled on a confined area or stored in a deep pit for longer than six months. In low-rainfall areas, the stockpile can be piled on the field with the installation of nutrient runoff control. In higher rainfall areas, a concrete pad and wall are constructed to store solid manure and prevent nutrient runoff from heavy rain.

Greenhouse gases generated from both storage methods have a pattern similar to that of enteric fermentation. Carbon and nitrogen compounds in manure are broken down by microbes to CH_4, and N_2O. The main factors influencing GHG emissions from storage are temperature and storage time. Due to the longer storage time, long-term stockpile solid manure storage generates a significant amount of GHGs. Temporary stack, as a short-term manure storage method, generates less GHGs than the long-term stockpile solid storage. However, it is still necessary to quantitatively delineate the emissions in order to assist livestock and poultry farms in evaluating their manure management operations. Temporary stack and long-term stockpiles of manure also produce NH_3; proposed methods to estimate NH_3 emissions are presented in Appendix 5-C.

The IPCC Tier 2 methodology is provided for estimating CH_4 emissions from temporary stacks or long-term stockpiles. This methodology uses a combination of IPCC and country-specific emission factors from the U.S. EPA GHG Inventory. The amount of manure, volatile solids content, and temperature are specific to the entity. The method for calculating N_2O emissions is the same as the equation presented in the U.S. GHG Inventory.

Rationale for Selected Method

The IPCC equations are the only available methods for estimating CH_4, and N_2O emissions from temporary stack and long-term stockpiles. These methodologies best describe the quantitative relationship among activity data at the entity level.

Activity Data

In order to estimate the daily CH_4 emissions, the following information is needed:[9]

- Animal type
- Total dry manure
- Volatile solids of dry manure[10]
- Temperatures (local ambient temperature and manure temperature)

In order to estimate the daily N_2O emission, the following information is needed:

- Total dry manure
- Total nitrogen content of the manure

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9: Total Nitrogen Entering Manure Storage and Treatment The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

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[9] Although daily estimates for the activity data are optimal, tracking this level of detail would be burdensome. Annual estimates don't allow for seasonal variation in diets and climate. Consequently, disaggregation of the data by season or by periods of major shifts in animal population is suggested.

[10] Volatile solids, total nitrogen content, and ammonia-nitrogen content should be obtained through sampling and lab testing.

Ancillary Data

The ancillary data used to estimate CH_4 emission for temporary storage and long term stockpiles are: maximum CH_4 producing capacities (B_0) and MCFs. The B_0 values for solid manure storage are obtained from the IPCC and listed in Table 5-19. Methane conversion factors for different manure management systems (including temporary storage of solid manure) are also obtained from the IPCC and listed in Table 5-20 and 5-16.

The ancillary data used to estimate N_2O emissions for temporary storage and long term stockpiles are the N_2O emission factors for solid manure storage systems are presented in Table 5-23 (U.S. EPA, 2011).

5.4.1.2 Method

Methane Emissions from Temporary Stack and Long-Term Stockpile

The Tier 2 approach by the IPCC model is recommended to estimate CH_4 emissions and is described in Equation 5-26 (IPCC, 2006). Daily CH_4 emission is estimated as a function of the volatile solids in manure placed into the storage and the animal-specific MCF.

Equation 5-26: IPCC Tier 2 Approach for Estimating CH_4 Emissions

$$\mathbf{E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{MCF}{100}}$$

Where:

- E_{CH4} = CH_4 emissions per day (kg CH_4 day^{-1})
- m = Total dry manure per day [a] (kg dry manure day^{-1})
- VS = Volatile solids (kg VS (kg dry manure)$^{-1}$)
- B_0 = Maximum CH_4 producing capacity for manure (m^3 CH_4 (kg VS)$^{-1}$)
- MCF = CH_4 conversion factor for the manure management system (%)
- 0.67 = Conversion factor of m^3 CH_4 to kg CH_4

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

Table 5-19: Maximum CH_4 Producing Capacities (B_0) from Different Animals

Animal	Maximum CH_4 Producing Capacity (B_0) (m^3/kg VS)	Animal	Maximum CH_4 Producing Capacity (B_0) (m^3/kg VS)
Beef replacement heifers	0.33[b]	Breeding swine	0.48
Dairy replacement heifers	0.17[b]	Layer (dry)	0.39
Mature beef cows	0.33[b]	Layer (wet)	0.39
Steers (>500 lbs)	0.33[b]	Broiler	0.36
Stockers (All)	0.17[b]	Turkey	0.36
Cattle on feed	0.33[b]	Duck	0.36
Dairy cow	0.24[b]	Sheep	0.19[b]
Cattle	0.19[b]	Feedlot sheep	0.36[b]
Buffalo	0.1[a]	Goat	0.17[b]
Market swine	0.48	Horse	0.3
		Mule/Ass	0.33

[a] There are no data for North America region; the data from Western Europe are used to calculate the estimation.
[b] Numbers are from the EPA U.S. Inventory: 1990-2009 (U.S. EPA, 2011). Other numbers are from IPCC (2006).

Table 5-20: Methane Conversion Factors for Temporary Storage of Solid Manure from Different Animals

Animal	Methane Conversion Factor (%)		
	Temp = 10-14°C	Temp = 15-25°C	Temp = 26-28°C
Dairy cow	1	1.5	2
Cattle	1	1.5	2
Buffalo	1	1.5	2
Market swine	1	1.5	2
Breeding swine	1	1.5	2
Layer (dry)	1.5	1.5	1.5
Broiler	1.5	1.5	1.5
Turkey	1.5	1.5	1.5
Duck	1	1.5	2
Sheep	1	1.5	2
Goat	1	1.5	2
Horse	1	1.5	2
Mule/Ass	1	1.5	2

Source: IPCC (2006).

Table 5-21: Methane Conversion Factors for Long-Term Stock Storage of Solid Manure from Different Animals

Animal	Methane Conversion Factor (%)		
	Temp = 10-14°C	Temp = 15-25°C	Temp = 26-28°C
Dairy cow	2	4	5
Cattle	2	4	5
Buffalo	2	4	5
Market swine	2	4	5
Breeding swine	2	4	5
Layer (dry)	1.5	1.5	1.5
Broiler	1.5	1.5	1.5
Turkey	1.5	1.5	1.5
Duck	1	1.5	2
Sheep	1	1.5	2
Goat	1	1.5	2
Horse	1	1.5	2
Mule/Ass	1	1.5	2

Source: IPCC (2006).

Table 5-22: Methane Conversion Factors for Long-Term Storage of Slurry Manure from Buffalo

Temperature (°C)	Methane Conversion Factor (%)
10	17
11	19
12	20
13	22
14	25
15	27
16	29
17	32
18	35
19	39

Temperature (°C)	Methane Conversion Factor (%)
20	42
21	46
22	50
23	55
24	60
25	65
26	71
27	78
28	80

Source: IPCC (2006).

Nitrous Oxide Emissions from Temporary Stack and Long-Term Stockpile

Nitrous oxide emissions are dependent on nitrification and denitrification. Manure storage is one of the main sources of U.S. overall N_2O emissions. The only quantitative method for estimating N_2O emissions from solid manure is the IPCC Tier 2 approach, which is also used for the U.S. Inventory. This approach is based on the use of emission factors from the most recent IPCC Guidelines and total nitrogen values are estimated according to Equation 5-9. Equation 5-27 presents the equation to estimate the N_2O emissions for solid manure.

Equation 5-27: IPCC Tier 2 Approach for Estimating N_2O Emissions

$$\mathbf{E_{N_2O} = m \times EF_{N2O} \times TN \times \frac{44}{28}}$$

Where:

E_{N2O} = Nitrous oxide emission per day (kg N_2O day^{-1})

m = Total dry manure per day[a] (kg dry manure day^{-1})

EF_{N2O} = N_2O emission factor (kg N_2O-N kg N^{-1})

TN = Total nitrogen at a given day (kg N (kg dry manure)$^{-1}$)

$\frac{44}{28}$ = Conversion of N_2O-N emissions to N_2O emissions

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

Table 5-23: N_2O Emission Factors for Solid Manure Storage

Type of Storage	N_2O Emission Factor (kg N_2O-N/kg N)
Temporary storage of solid/slurry manure	0.005
Long-term storage of solid manure	0.002
Long-term storage of slurry manure	0.005

Source: U.S. EPA (2011).

5.4.2 Composting

5.4.2.1 Overview of Composting

Method for Estimating Emissions from Manure Storage and Treatment –Composting

Methane

- IPCC Tier 2 approach, utilizing monthly data on volatile solids and dry manure. Volatile solids content can be obtained from sampling and lab testing.
- Method is the only readily available method.

Nitrous Oxide

- IPCC Tier 2 approach, utilizing data on a nitrous oxide emission factor, total initial nitrogen, and dry manure.
- Method depends on whether the system is in vessel, static pile, intensive windrow, or passive windrow.
- Method is only readily available method.

Composting is the controlled aerobic decomposition of organic material into a stable, humus-like product (USDA NRCS, 2007). Animal manure may be composted in a variety of different systems, including in-vessel systems, windrows, or static piles. In-vessel systems handle compost in a closed system such as a rotary drum or box that incorporates regular movement to ensure proper aeration. The largest composting operations divide up the compost into long heaps for windrow composting or into one large pile for aerated static pile composting. In the former method, proper oxygen flow can be maintained via manual turning or pipe systems, whereas in the latter method, it is maintained through pipe systems. Composting has become a popular method in some regions to decrease the volume and weight of livestock manure and to produce a product that is often more acceptable to farmers as a fertilizer. During a 100- to 120-day composting period, the weight and volume of manure may be decreased by 15 to 70 percent (Eghball et al., 1997; Inbar et al., 1993; Lopez-Real & Baptista, 1996). Furthermore, the heat generated through the composting process can kill parasites, pathogens, and weed seeds found in animal waste, creating a safer product for crop application.

The quantity of GHG emissions is affected by the composting method employed. Hao et al. (2001) reported that GHG emissions from cattle manure compost increased about twofold when the compost was actively composted rather than passively composted in windrows. Active windrows were turned six times (days 14, 21, 29, 50, 70, and 84). Passive windrows were never turned, but air was introduced into the windrows by a series of open-ended perforated steel pipes. To the extent that the rate of GHG formation depends on oxygen saturation in the pore space, aeration method (i.e., forced-air vs. passive/convective) and rate (or turning frequency) will affect the magnitude of GHG emissions during the composting process.

Eghball et al. (1997) reported that 19 to 45 percent of the nitrogen present in manure was lost during composting, with the majority of this presumably as NH_3. Using changes in the nitrogen:phosphorus ratio of feedlot manure that was placed in compost windrows and the nitrogen:phosphorus ratio of “finished” compost, Cole et al. (2011) estimated that 10 to 20 percent of nitrogen was lost during composting. The U.S. EPA currently assumes that one to 10 percent of nitrogen entering compost systems is lost as N_2O (IPCC, 2006; U.S. EPA, 2009).

The IPCC Tier 2 methodology is provided for estimating CH_4 and N_2O emissions from composting. This methodology uses country-specific emission factors from the U.S. EPA GHG Inventory. The amount of manure, volatile solids content, and temperature are specific to the entity. The GHG estimation method for manure composting does not consider other organic carbon sources that might be added into manure composting.

Rationale for Selected Method

The IPCC equations are the only available methods for estimating CH_4 and N_2O emissions from composting. These methodologies best describe the quantitative relationship amongst activity data at the entity level.

5.4.2.2 Activity Data

In order to estimate the daily CH_4 emissions, the following information is needed:

- Animal type
- Total dry manure
- Volatile solids of dry manure
- Temperatures (local ambient temperature and manure temperature)

In order to estimate the daily N_2O emissions, the following information is needed:

- Total dry manure in the storage

- Total nitrogen in manure

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9: Total Nitrogen Entering Manure Storage and Treatment The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

5.4.2.3 Ancillary Data

The ancillary data used to estimate CH_4 emissions for manure composting are: maximum CH_4 producing capacities (B_0) and MCFs. The B_0 values are obtained from the IPCC (2006) and listed in Table 5-19. The MCF values are obtained from EPA (U.S. EPA, 2011) and listed in Table 5-24.

The ancillary data used to estimate N_2O emission for manure composting are the N_2O emission factors (Table 5-25).

5.4.2.4 Method

Methane Emissions from Composting

The Tier 2 approach in the IPCC model is adapted with country-specific factors to estimate CH_4 emissions from composting of solid manure. Daily CH_4 emissions are estimated as a function of the volatile solids in manure placed into the storage and the MCF.

Equation 5-28: IPCC Tier 2 Approach for Calculating Methane Emissions from Composting Solid Manure

$$\mathbf{E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{MCF}{100}}$$

Where:

E_{CH4} = Methane emissions per day (kg CH_4 day^{-1})

m = Total dry manure[a] (kg dry manure day^{-1})

VS = Volatile solids (kg VS $(kg\ dry\ manure)^{-1}$)

B_0 = Maximum CH_4 producing capacity for manure (m^3 CH_4 $(kg\ VS)^{-1}$) (see Table 5-24)

MCF = Methane conversion factor for the manure management system (%)

0.67 = Conversion factor of m^3 CH_4 to kg CH_4

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

The B_0 values for composting solid manure are obtained from the IPCC (2006) and are listed in Table 5-19. Methane conversion factors for different approaches of composting solid manure are obtained from IPCC (2006).

Table 5-24: Methane Conversion Factors for Composting Solid Manure

Animal	Methane Conversion Factor (%)		
	Cool Climate	Temperate Climate	Warm Climate
Manure composting – in vessel	0.5	0.5	0.5
Manure composting – static pile	0.5	0.5	0.5
Manure composting – intensive windrow	0.5	1	1.5
Manure composting – passive windrow	0.5	1	1.5

Source: IPCC (2006).

Nitrous Oxide Emissions from Composting

A Tier 2 IPCC model is adapted to estimate N_2O emissions from composting of solid manure. Equation 5-29 presents the equation for estimating N_2O emissions from composting of solid manure. Emission factors for different composting methods are listed in Table 5-25 and total nitrogen is estimated according to Equation 5-9.[11]

Equation 5-29: IPCC Tier 2 Approach for Estimating N_2O Emissions from Composting of Solid Manure

$$\mathbf{E_{N_2O} = m \times EF_{N2O} \times TN \times \frac{44}{28}}$$

Where:

E_{N2O} = Nitrous oxide emissions per day (kg N_2O day^{-1})

m = Total dry manure[a] (kg day^{-1})

EF_{N2O} = N_2O emission (loss) relative to total nitrogen in manure (kg N_2O-N (kg TN)$^{-1}$)

TN = Total nitrogen in the initial (fresh) manure (kg TN (kg dry manure)$^{-1}$)

$\frac{44}{28}$ = Conversion of N_2O-N emissions to N_2O emissions

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

Table 5-25: N_2O Conversion Factors (EF_{N2O}) for Composting Solid Manure

Category	N_2O Emission Factor (kg N_2O-N/ kg TN)
Cattle and Swine Deep Bedding (Active Mix)	0.07
Cattle and Swine Deep Bedding (No Mix)	0.01
Pit Storage Below Animal Confinements	0.002

Source: IPCC (2006).

[11] Some studies have been conducted on the rate of N_2O emissions for swine (Fukummoto et al., 2003; Szanto et al., 2006) but this data is limited and further research is necessary. See Section 0 Research Gaps for further discussion.

5.4.3 Aerobic Lagoon

5.4.3.1 Overview of Aerobic Lagoons

Method for Estimating Emissions from Manure Storage and Treatment – Aerobic Lagoon

Methane

- The MCF for aerobic treatment is negligible and is designated as zero percent in accordance with the IPCC Guidance.

Nitrous Oxide

- IPCC Tier 2 method utilizing IPCC emission factors.
- Method takes into account the volume of the lagoon and the total nitrogen content of the manure.
- Method is the only readily available method.

Aerobic lagoons are man-made outdoor basins that hold animal wastes. The aerobic treatment of manure involves the biological oxidation of manure as a liquid, with either forced or natural aeration. Natural aeration is limited to aerobic lagoons with photosynthesis and is consequently shallow to allow for oxygen transfer and light penetration. These systems become anoxic during low-sunlight periods. Due to the depth limitation, naturally aerated aerobic lagoons have large surface area requirements and are impractical for large operations.

The IPCC Tier 2 methodology is provided for estimating CH_4 and N_2O emissions from aerobic lagoons. This methodology uses a combination of IPCC and country-specific emission factors from the U.S. EPA GHG Inventory. Aerobic conditions result in the oxidation of carbon to CO_2, not the reduction of carbon to CH_4, thus CH_4 emissions from aerobic lagoons is considered negligible and is designated as zero in accordance with IPCC. The method for calculating N_2O emissions accounts for the volume of the lagoon as well as the total nitrogen content of the manure.

5.4.3.2 Rationale for Selected Methods

The IPCC equations are the only available methods for estimating CH_4, and N_2O emissions from aerobic lagoons. These methodologies best describe the quantitative relationship among activity data at the entity level.

5.4.3.3 Activity Data

No activity data are needed (MCF=0) for the estimation of CH_4 gas emissions.

In order to estimate the daily N_2O emissions, the following information is needed:

- Surface area of lagoon
- Volume of the material in the lagoon
- Total nitrogen content of the manure

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9. The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

5.4.3.4 Ancillary Data

The ancillary data used to estimate N_2O emissions for aerobic lagoon are N_2O emission factors (U.S. EPA, 2011).

5.4.3.5 Method

Methane Emissions from Aerobic Lagoon

The MCF for aerobic treatment is negligible and was designated as zero percent in accordance with the IPCC (2006). The solids from the bottom of the lagoon have significant volatile solids and B_0 associated with livestock type; the characteristics of the solids should be measured and used as the inputs to estimate emissions of GHGs for subsequent storage and treatment operations.

Nitrous Oxide Emissions from Aerobic Lagoon

The Tier 2 approach in the IPCC model is adapted to estimate N_2O emissions from aerobic lagoons. The N_2O conversion factors for different aeration system are listed in Table 5-26. The estimation method for N_2O emissions is provided in Equation 5-30.

Table 5-26: N_2O Conversion Factors (EF_{N2O}) for Aerobic Lagoons

Aeration Type	N_2O Conversion Factor (kg N_2O-N/kg N)
Natural aeration	0.01
Forced aeration	0.005

Source: IPCC (2006).

Equation 5-30: Calculating N_2O emissions from Aerobic Lagoons

$$E_{N_2O} = V \times EF_{N2O} \times TN \times \frac{44}{28}$$

Where:

E_{N2O} = Nitrous oxide emissions per day (kg N_2O day^{-1})

V = Total volume of the lagoon liquid (m^3 day^{-1})

EF_{N2O} = Nitrous oxide emission (loss) relative to total nitrogen in the lagoon liquid (kg N_2O-N (kg TN)$^{-1}$)

TN = Total nitrogen in the lagoon liquid (kg TN m^{-3})

$\frac{44}{28}$ = Conversion of N_2O-N emissions to N_2O emissions

5.4.4 Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks

5.4.4.1 Overview of Anaerobic Lagoons, Runoff Holding Ponds, and Storage Tanks

Method for Estimating Emissions from Manure Storage and Treatment – Anaerobic Lagoons, Runoff Holding Ponds, Storage Tanks

Methane

- Sommer model (Sommer et al., 2004) is used with degradable and nondegradable fractions of volatile solids from Møller et al. (2004).
- This method was selected as it accounts for manure temperature and total volatile solids content of manure. Volatile solids content can be obtained from sampling and lab testing.

Nitrous Oxide

- Emissions are a function of the exposed surface area and U.S.-based emission factors.
- Method is the only readily available option.

The most frequently used liquid manure storage systems are anaerobic lagoons (in the Southern portion of the United States), earthen or earthen-lined storages (in the Northern portion of the country), runoff holding ponds, and above-grade storage tanks. Anaerobic lagoons are earthen basins that provide an environment for anaerobic digestion and storage of animal waste. Both the American Society of Agricultural and Biological Engineers and U.S. Department of Agriculture Natural Resources Conservation Service have engineering design standards for construction and operation of anaerobic lagoons. In most feedlots a holding pond is constructed to collect runoff for short-term storage. Storage tanks range from lower-cost earthen basins to higher-cost, glass-lined steel tanks. The manure that enters these systems is usually diluted with flush water, water wasted at stalls, and rainwater.

All of these storage systems (without aeration) are biologically-anaerobic lagoons, which mean that they have similar potential, as with enteric fermentation, to produce CH_4 and N_2O. Due to the large quantity of liquid manure produced in the United States, liquid manure storage can be a major source of GHG emissions from animal operations. In terms of estimation of GHG emission from anaerobic lagoon/runoff holding pond/storage tanks, these storage systems are classified into four categories: 1) covered storage with a crust formed on the surface; 2) covered storage without a crust formed on the surface; 3) uncovered storage with a crust formed on the surface; and 4) uncovered storage without a crust formed on the surface.

The algorithms for calculating CH_4 emissions described by Sommer et al. (2004) are recommended for estimating emissions at the entity-level. The model considers volatile solids to be the main factor influencing emissions from manure and relates emissions to the content of degradable volatile solids. Nitrous oxide is estimated as a function of the exposed surface area of the manure storage and whether a crust is present on the surface.

Rationale for Selected Methods

The Sommer algorithms link carbon turnover, volatile solids, temperature, and storage time to CH_4 emissions estimates and is the best available method for estimating CH_4 emissions at the entity level. The method provided for N_2O is the only available method for estimating emissions. These methodologies best describe the quantitative relationship among activity data at the entity level.

5.4.4.2 Activity Data

In order to estimate the daily CH_4 emissions, the following information is needed:

- Animal type
- Total dry manure
- Volatile solids in the storage
- Temperatures (local ambient temperature and manure temperature)

In order to estimate the N_2O emission, the following information is needed:

- Total dry manure
- Total nitrogen content of the manure
- The exposed surface area of the manure storage

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9. The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

5.4.4.3 Ancillary Data

The ancillary data used to estimate CH_4 emissions for anaerobic lagoons, runoff holding ponds, and storage tanks are the maximum CH_4 producing capacities (B_0), potential CH_4 yield ($E_{CH4,\,pot}$), rate correcting factors (b_1 and b_2), Arrhenius constant (A), activation energy (E), gas constant (r), and collection efficiency (η) for liquid manure storage from different animals. These data are available from the IPCC (2006) and Sommer et al. (2004) and are listed in Table 5-27.

The ancillary data used to estimate N_2O emissions for anaerobic lagoons, runoff holding ponds, and storage tanks is the N_2O emission factor from Table 5-29 (U.S. EPA, 2011).

5.4.4.4 Method

Methane Emissions from Anaerobic Lagoons, Runoff Holding Ponds, Storage Tanks

The Sommer model (Sommer et al., 2004) is used as the estimation method for CH_4 emission (Rotz et al., 2011b). Daily CH_4 emissions are estimated as a function of manure temperature and the volatile solids in manure placed into liquid storages. The parameters for the estimation are listed in Table 5-28.

Equation 5-31: Using the Sommer Model to Calculate Daily CH_4 Emissions

$$\mathbf{E_{CH_4} = m \times 0.024 \times (VS_d \times b_1 + VS_{nd} \times b_2) \times e^{\ln(A) - \frac{E}{RT}} \times (1 - \eta)}$$

Where:

E_{CH4}	= Methane emission per day (kg CH_4 day^{-1})
m	= Total dry manure per day (kg dry manure day^{-1})[a]
0.024	= Dimensionless factor to modify the Sommer model based on VS
VS_d and VS_{nd}	= Degradable and nondegradable VS in the manure, respectively (kg (kg dry manure)$^{-1}$)
b_1 and b_2	= Rate correcting factors (dimensionless)
A	= Arrhenius parameter (g CH_4 (kg VS)$^{-1}$ hr^{-1})
E	= Activation energy (J mol^{-1})
R	= Gas constant (J K^{-1} mol^{-1})
T	= Storage temperature (K)
η	= Collection efficiency of different liquid storage categories

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

The degradable fraction of the volatile solids is dependent on the potential CH_4 yield and the maximum CH_4 producing capacities and can be calculated using Equation 5-32. The fraction of nondegradable volatile solids (material that is not broken down by microorganisms) is calculated from the total volatile solids content and degradable fraction of the volatile solids, as described by Equation 5-33. The B_0 values are obtained from the IPCC (2006) and are listed in Table 5-19.

Equation 5-32: Calculating the Degradable Fraction of the Volatile Solids

$$\mathbf{VS_d = VS_T \times \frac{B_0}{E_{CH4,pot}}}$$

Where:

VS_d = Degradable VS fractions in the manure on a given day (kg (kg dry manure)$^{-1}$)

VS_T = Volatile solids content in the storage on a given day (kg (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacities (kg CH_4 (kg VS)$^{-1}$)

$E_{CH4,\,pot}$ = Potential CH_4 yield of the manure (kg CH_4 (kg VS)$^{-1}$)

Equation 5-33: Calculating the Non-Degradable Fraction of the Volatile Solids

$$\mathbf{VS_{nd} = VS_T - VS_d}$$

Where:

VS_d and VS_{nd} = Degradable and nondegradable VS fractions in the manure on a given day (kg (kg dry manure)$^{-1}$), respectively

VS_T = Volatile solids content in the storage on a given day (kg (kg dry manure)$^{-1}$)

The collection efficiency (η) depends on different liquid storage categories of: 1) covered storage with a crust formed on the surface; 2) covered storage without a crust formed on the surface; 3) uncovered storage with a crust formed on the surface; and 4) uncovered storage without a crust formed on the surface. A crust allows air and CH_4 to be retained on the surface of the manure storage and increases the potential for oxidation of CH_4 (Hansen et al., 2009; Nielsen et al., 2010). When a crust does not form, CH_4 is directly emitted without rapid oxidation. For cattle slurry and pig slurry, degradable and nondegradable volatile solids (as a fraction of VS_T) are given in Table 5-28.

Table 5-27: Parameters for Estimating CH_4 Emission from Liquid Manure Storage

Parameters		Cattle	Swine
Arrhenius constant (ln(A)) – g CH_4 (kg VS)$^{-1}$ hr^{-1}		43.33	43.21
Activation energy (E) – J mol^{-1}		1.127×10^5	1.127×10^5
Gas constant (R) – J K^{-1} mol^{-1}		8.314	8.314
Rate correction factor for VS_d (b_1)		1	1
Rate correction factor for VS_{nd} (b_2)		0.01	0.01
Potential methane yield of the manure ($E_{CH4,\,pot}$) (kg CH_4/ kg VS)		0.48	0.50
Collection efficiency (η)	Covered storage with a crust form on the surface[a]	1	1
	Covered storage without a crust form on the surface[a]	1	1
	Uncovered storage with a crust form on the surface[b]	0	0
	Uncovered storage without a crust form on the surface[c]	-0.4	-0.4

Source: Sommer et al. (2004) and IPCC (2006).

[a] CH_4 gas from covered storage with a crust form on the surface is collected and flared.

[b] Uncovered storage with a crust form on the surface is used for the derivation of Equation 5-22.

[c] The emission for uncovered storage without a crust is 40 percent greater than uncovered storage with a crust, so the collection efficiency for this case is -40 percent.

Table 5-28: Degradable and Nondegradable Volatile Solids for Cattle and Swine Manure

Type of Manure	VS_d/VS_T	VS_{nd}/VS_T
Cattle liquid manure	0.46	0.54
Swine liquid manure	0.89	0.11

Source: Møller et al. (2004).

Nitrous Oxide Emissions from Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks

Nitrous oxide emissions from liquid manure storage typically represent a relatively small portion of the N_2O emissions from farms. Most studies indicate the criticality of the crust for the formation and emission of N_2O (Petersen and Sommer, 2011). Therefore, N_2O emissions from liquid manure storage are estimated as a function of the exposed surface area of the manure storage and the presence of a crust on the surface.

Equation 5-34: Calculating N_2O Emissions from Liquid Manure Storage

$$\mathbf{E_{N_2O} = EF_{N2O} \times \frac{A_{surface}}{1000}}$$

Where:

E_{N2O} = Nitrous oxide emissions per day (kg N_2O day^{-1})

EF_{N2O} = Emission rate of N_2O (g N_2O m^{-2} day^{-1})

$A_{surface}$ = Exposed surface area of the manure storage (m^2)

1,000 = Conversion factor for grams to kilograms $\left(\frac{1\ \text{kg}}{1000\ \text{g}}\right)$

The emission factor of N_2O is dependent on crust formation on the liquid storage. The crust allows air to be retained on the surface of the manure storage and increases the potential for nitrification and denitrification (Hansen et al., 2009; Nielsen et al., 2010). When a crust does not form, oxygen is not retained on the liquid surface with nitrogenous compounds, and therefore no N_2O is formed and emitted. The emission factors of N_2O for different liquid storage methods are listed in Table 5-29.

Table 5-29: Emission Factor of N_2O for Liquid Storage with Different Crust Formation

Type of Liquid Storage	$EF_{N2O,man}$ (g $N_2O/m^2/day$)
Uncovered liquid manure with crust	0.8
Uncovered liquid manure without crust	0
Covered liquid manure	0

Source: Rotz et al. (2011a).

5.4.5 Anaerobic Digester with Biogas Utilization

5.4.5.1 Overview of Anaerobic Digester with Biogas Utilization

Method for Estimating Emissions from Manure Storage and Treatment – Anaerobic Digester with Biogas Utilization

Methane

- IPCC Tier 2 using Clean Development Mechanism EFs for digester types to estimate CH_4 leakage from digesters.
- Anaerobic digester systems convert organic matter in manure into CH_4 and subsequently combust CH_4 into CO_2.
- Gas leakage from digesters is the main source of GHG emission.
- Leakage of CH_4 from the anaerobic digester system is estimated.

Nitrous Oxide

- N_2O leakage from digesters is fairly small and negligible.

One of the most commonly discussed waste management alternatives for GHG reduction and energy generation is anaerobic digestion. Anaerobic digestion is a natural, biological conversion process that has been proven effective at converting wet organic wastes into biogas (approximately 60 percent CH_4 and 40 percent CO_2). Biogas can be used as a fuel source for engine-generator sets, producing relatively clean electricity while also reducing some of the environmental concerns associated with manure. The digester can be as simple as a covered anaerobic lagoon (Gould-Wells and Williams, 2004) or as sophisticated as thermophilic or media matrix (attached growth) digesters (Cantrell et al., 2008a). There are a wide variety of anaerobic digestion configurations, such as continuous stirred tank reactor (CSTR), covered lagoon, plug-flow, temperature phased, upflow anaerobic sludge blanket (UASB), packed-bed, and fixed film. The digestion is also categorized based on culture temperature: thermophilic digestion in which manure is fermented at a temperature of around 55°C, or mesophilic digestion at a temperature of around 35°C. Among these technologies, CSTR, plug-flow, and covered lagoon, all under mesophilic conditions, are the most often-used methods.

During anaerobic digestion, a group of microbes work together to convert organic matter into CH_4, CO_2, and other simple molecules. The main advantages of applying anaerobic digestion to animal manures are odor reduction, electricity generation, and the reduction of GHG emissions and manure-borne pathogens. Anaerobic digestion is also an excellent pre-treatment process for subsequent manure treatment to remove organic matter and concentrate phosphorus. Considering the small amount of N_2O existing in biogas, N_2O emissions are not estimated for the anaerobic digestion of liquid manure.

The challenges associated with anaerobic digestion relate to initial capital cost, operation, and maintenance and other gases that may be generated (e.g., nitric oxides). The economics relate to access to the electrical grid and sufficient green-electricity offsets to make the operation profitable. Profitable conditions are relatively scarce. Finally, the digester sludge must be managed. Another conversion alternative with energy creation potential is thermochemical conversion (Cantrell et al., 2008a). Systems that use thermochemical conversions to syngases, bio-oil, and biochar for electricity and fuel are emerging, but are not yet established.

Since an anaerobic digestion system converts organic carbon in manure into CH_4 and subsequently combusts CH_4 into CO_2, the GHG emissions from manure anaerobic digestion operation are mainly from the leakage of digesters. The leakage of CH_4 can be estimated based on the IPCC Tier 2 approach in combination with technology-specific emission factors.

5.4.5.2 Rationale for Selected Method

The IPCC equation is the only available method for estimating CH_4 emission from digesters. This methodology best describes the quantitative relationship among activity data at the entity level and takes into account the specific technology employed.

5.4.5.3 Activity Data

In order to estimate the CH_4 leakage from anaerobic digestion, the following information is needed:

- Animal type
- Total dry manure into the digester
- Volatile solids in the manure
- Digester temperatures

5.4.5.4 Ancillary Data

Ancillary data for anaerobic digestion effluent are needed for further estimation of CH_4 and N_2O emissions from post-treatment approaches such as aerobic or anaerobic lagoons, nutrient removal operations, etc. Thus, the necessary data for the effluent include effluent flow rate, total solids, volatile solids, chemical oxygen demand, effluent temperature, environmental temperature, liquid/solid separation methods, and total nitrogen.

5.4.5.5 Method

Equation 5-35 describes the IPCC Tier 2 approach for estimating CH_4 emissions for anaerobic digesters. The CH_4 generated from digesters is assumed to be flared or used as a biogas; the only emissions from digesters are from system leakage.

Equation 5-35: IPCC Tier 2 Approach for Estimating CH_4 Emissions

$$\mathbf{E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{EF_{CH_4,leakage}}{100}}$$

Where:

E_{CH4}	= CH_4 emissions per day (kg CH_4 day^{-1})
m	= Total dry manure per day (kg day^{-1})
VS	= Volatile solids (kg VS (kg dry manure)$^{-1}$)
B_0	= Maximum CH_4 producing capacity for manure from different animal (m^3 CH_4 (kg VS)$^{-1}$)
0.67	= Conversion factor from weight to volume of methane (kg CH_4 m^{-3})
$EF_{CH4,\ leakage}$	= Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester (%)

The B_0 values are obtained from the IPCC (2006) and are listed in Table 5-19. The emission factors for the amount of CH_4 leakage by technology are listed in Table 5-30.

Table 5-30: Emission Factors for the Fraction of Methane Leaking from Digesters

Digester Configurations	$EF_{CH4, leakage}$ (%)
Digesters with steel or lined concrete or fiberglass digesters with a gas holding system (egg shaped digesters) and monolithic construction	2.8
UASB type digesters with floating gas holders and no external water seal	5
Digesters with unlined concrete/ferrocement/brick masonry arched type gas holding section; monolithic fixed dome digesters	10
Other digester configurations	10

Source: CDM (2012).

5.4.6 Combined Aerobic Treatment Systems

Method for Estimating Emissions from Combined Aerobic Treatment Systems

- Method is to utilize 10 percent of the emissions resulting from estimation of emissions from Liquid Manure Storage and Treatment – Anaerobic Lagoons, Runoff Holding Ponds, and Storage Tanks.
- Method based on research findings that systems avoid 90 percent of the GHG emissions from standard anaerobic lagoon treatment.

Dealing with the total treatment of wastewater from either swine or dairy is complex, because the liquid and solid phases must be treated. In municipal sewage treatment systems, the wastewater is very dilute so the treatment of the biochemical oxygen demand by aeration is a fundamental process. In contrast, the solids content of livestock wastewater is quite high, as is the biochemical oxygen demand. Consequently, the cost of stabilizing the biochemical oxygen demand with aeration has proven to be uneconomical. A successful solution to this problem was developed by Vanotti et al. (2007), who used polyacrylamide flocculation to remove more than 90 percent of the solids (Vanotti and Hunt, 1999; Vanotti et al., 2002). The solid fraction was then composted (Vanotti, 2006). The remaining liquid was transferred to a separated water tank where it was subsequently aerated (Vanotti and Hunt, 2000; Vanotti et al., 2007; Vanotti and Szogi, 2008). During these two phases of treatment, more than 90 percent of the GHG emissions from standard anaerobic lagoon treatment were avoided (Vanotti et al., 2008). The avoidance was achieved by aerobic treatment of the solids via composting and nitrification/denitrification in the liquid effluent.

After nitrification/denitrification, the treated effluent moves to the settling tank and subsequently into the phosphorus treatment chamber. Here the wastewater, which has low alkalinity, is amended with liquid lime, and the pH is raised to approximately 10. In the presence of high pH and calcium, the phosphorus is precipitated and the pathogens are killed (Vanotti et al., 2003; Vanotti et al., 2005; Vanotti et al., 2009). The treated wastewater is then recycled into the houses. This process provides a healthier environment for the pigs (Vanotti et al., 2009). The system must be operated to ensure proper and timely flushing of the house. The polyacrylamide addition and the solids separation units must be operated properly. Aeration of the nitrification tank must be maintained, as must the addition of liquid lime. The pumps that maintain the internal recycling must also be maintained and operated correctly. This system is the only treatment system to meet and be certified for expansion of swine production in North Carolina.

To estimate emissions for combined aerobic treatment systems, the methodology for anaerobic lagoons, runoff holding ponds, and storage tanks is applied to the system. Gas emissions of CH_4 and N_2O are estimated using 10 percent of the values for emissions from anaerobic lagoon treatment.

5.4.7 Sand-Manure Separation

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Sand/Manure Separation

- No method is provided as GHG emissions are negligible from the sand/manure separation process. However, resulting volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid manure should be measured and used as the inputs to estimate emissions of GHGs for subsequent storage and treatment operations.

Sand is one of the standard materials for dairy cow bedding. It provides superior cow comfort, environment for udder health (and consequently better milk quality), and traction when compared with organic bedding materials. Sand separation systems can be classified as mechanical separation and sedimentation separation. Sedimentation separation uses dilution water and gravity to allow sand to passively settle in sand traps. Due to the high organic material content contained in the settled sand, the sand recovered from the sand trap needs to be drained multiple times and dried prior to reuse. Mechanical sand-manure separation systems use recycled liquid manure and aeration to suspend manure solids, settle sand at the bottom of the separator, and recover the sand using a heavy duty auger. Sand is generally discharged with less than two percent organic matter. The mechanically separated sand can be reused for bedding.

Since sand-manure separation is relatively quick (compared with other storage and treatment methods), GHG emissions from the operation are minimal. The process of separating sand and manure is not assumed to contribute to GHG emissions. After sand-manure separation, the separated liquid manure is treated as the influent for the next step of storage and treatment operations. The various storage and treatment operation options are shown in Figure 5-7. The parameters of volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid manure should be measured, and used as the inputs to estimate emissions of GHGs.

5.4.8 Nutrient Removal

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Nutrient Removal

- Not estimated due to limited quantitative information on GHGs from nitrogen removal processes.

Nitrogen and phosphorus are the primary elements that cause eutrophication in surface waters. With increased Federal, State and local attention on non-point waste sources, more and more animal operations will likely use nutrient removal approaches to treat liquid manure before land application and other uses. Compared to phosphorus, nitrogen in manure contributes to N_2O emission; removing it can significantly alleviate emissions. Nitrogen in manure comprises NH_3, particulate organic nitrogen, and soluble organic nitrogen. Five main nitrogen removal approaches—Biological Nitrogen Removal (BNR), Anamox, NH_3 stripping, ion exchange, and struvite crystallization—have been applied for municipal and industrial wastewater, as well as for animal waste streams. Because N_2O originates from nitrogen sources, quantification of nitrogen removal is important to estimate emissions from animal manure.

Because most nitrogen removal methods for liquid manure are currently in the research and development stage, very little quantitative information is available on the nitrogen removal methods mentioned above for animal manure under different operation conditions. The suggested estimation method is to consider the liquid manure after nutrient removal as the influent for storage and treatment approaches that entities will use to further treat liquid manure. Measurements of volatile solids, total nitrogen, organic nitrogen, and manure temperature of the treated liquid manure are needed to estimate CH_4 and N_2O emissions.

5.4.9 Solid–Liquid Separation

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Solid– Liquid Separation

- No method is provided as GHG emissions are negligible. However, resulting volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid and solid manure should be measured and used as the inputs to estimate emissions of GHGs and NH_3 for subsequent storage and treatment operations.

Solid–liquid manure separation has been used widely by dairy farms. One purpose of solid–liquid separation is to physically separate and remove the larger solids from liquid manure in order to store and treat them separately. The available commercial methods include gravity sedimentation and mechanical separation (with or without coagulation flocculation). Sedimentation and mechanical separation without coagulation flocculation are the most popular methods used by animal farms. Similar to sand–liquid manure separation, GHG emissions from the operation are minimal; however, separation has an impact on nutrient distribution in separated solid and liquid manure, which will influence GHG emissions from the next stage of manure storage and treatment for solid and liquid manure. The separated liquid manure is treated as the influent for the next step of storage and treatment operations. The possible storage and treatment options are delineated in Figure 5-7.

The parameters of total solids (dry manure), total nitrogen, organic nitrogen, and manure temperature of the separated liquid and solid manure should be measured, and used as the inputs to estimate GHGs emission in the subsequent storage and treatment operations. The distribution of total solids after solid–liquid separation for typical mechanical separators are listed in Table 5-317 (Ford and Fleming, 2002).

Table 5-31: Efficiency of Different Mechanical Solid-Liquid Separation

Separation Technique	Manure Type	Screen Size (mm)	Influent (% DM)	Total Solid Removal Efficiency (%)	Source
Screen					
Stationary inclined screen	Swine	1.0	0.0-0.7	35.2	Shutt et al. (1975)
	Beef	0.5	0.97-4.41	1-13	Hegg et al. (1981)
	Dairy	1.5	3.83	60.9	Chastain et al. (2001)
Vibrating screen	Swine	0.39	0.2-0.7	22.2	Shutt et al. (1975)
	Beef	0.52-1.91	5.5-7.4	4-44	Gilbertson and Nienaber (1978)

Separation Technique	Manure Type	Screen Size (mm)	Influent (% DM)	Total Solid Removal Efficiency (%)	Source
	Beef	0.64-1.57	1.55-3.19	6-16	Hegg et al. (1981)
	Dairy	0.64-1.57	0.95-1.9	8-16	Hegg et al. (1981)
	Swine	0.64-1.57	1.55-2.88	3-27	Hegg et al. (1981)
	Swine	0.10-2.45	1.5-5.4	11-67	Holmberg et al. (1983)
Rotating screen	Beef	0.75	1.56-3.68	4-6	Hegg et al. (1981)
	Dairy	0.75	0.52-2.95	0-14	Hegg et al. (1981)
	Swine	0.75	2.54-4.12	4-8	Hegg et al. (1981)
In-channel flighted conveyor screen	Dairy	3	7.1	4.22	Møller et al. (2000)
	Swine	3	5.66	25.8	Møller et al. (2000)
Centrifugal					
Centrifuge	Beef		7.5	25	Glerum et al. (1971)
Centrisieve	Swine		5-8	30-40	Glerum et al. (1971)
Decanter centrifuge	Beef		6.9	64	Chiumenti et al. (1987)
	Beef		6.0	45	Chiumenti et al. (1987)
	Swine		7.58	66	Glerum et al. (1971)
	Swine		1.9-8.0	47.4-56.2	Sneath et al. (1988)
Liquid cyclone	Swine			26.5	Shutt et al. (1975)
Filtration/pressing					
Roller press	Swine		5.2	17.3	Pos et al. (1984)
	Dairy		4.8	25	Pos et al. (1984)
	Beef		4.5	13.3	Pos et al. (1984)
Belt press	Dairy	1-2	7.1	32.4	Møller et al. (2000)
	Swine	1-2	5.7	22.3	Møller et al. (2000)
Screw press	Swine		5	16	Chastain et al. (1998)
	Swine		1-5	15-30	Converse et al. (1999)
	Dairy		1-10	15.8-47	Converse et al. (1999)
	Dairy		2.6	23.8	Converse et al. (1999)
	Dairy		4.9	33.4	Converse et al. (1999)
Fournier rotary press[a]	Swine			85	Ford and Fleming (2002) Fournier (2010)
Rotary vacuum filter	Swine		7.5	51	Glerum et al. (1971)
Pressure filter	Beef		7	76	Chiumenti et al. (1987)
Continuous Belt Microscreening Unit					
	Swine		2-8	40-60	Fernandes et al. (1988)

[a] With polymer addition.

5.4.10 Constructed Wetland

Globally, constructed wetlands are used for the treatment of wastewaters, capture of sediments,

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Constructed Wetland

- Currently no method is provided to estimate gas emission from constructed wetland of animal manure, although GHG sinks are noted to likely be greater than CH_4 and N_2O emissions, which are considered negligible.

and drainage water abatement (Hammer, 1989; Kadlec and Knight, 1996; Tanner et al., 1997; Hunt et al., 2002; Hunt et al., 2003; Picek et al., 2007; Harrington and McInnes, 2009; Mustafa et al., 2009; Soosaar et al., 2009; Elgood et al., 2010; Harrington and Scholz, 2010; VanderZaag et al., 2010; Chen et al., 2011; Locke et al., 2011; Tanner and Headley, 2011; Tanner and Sukias, 2011; Vymazal, 2011). Constructed wetlands are generally classified as sub-surface or surface flow wetlands (Kadlec and Knight, 1996). The sub-surface wetlands typically consist of wetland plants growing in a bed of highly porous media, such as gravel or wood chips. They are commonly used to improve drainage water quality. These wetlands are generally rectangular in shape and one to two meters in depth. There is lack of agreement about the relative impact of microbial and plant processes in the function of subsurface wetlands, including GHG production and emissions. However, it is accurate to say that plants and microbes are typically interdependently involved (Picek et al., 2007; Zhu et al., 2007; Wang et al., 2008; Faubert et al., 2010; Lu et al., 2010; Tanner and Headley, 2011). The microbial community advances biogeochemical processes (Tanner et al., 1997; Hunt et al., 2003; Zhu et al., 2007; Dodla et al., 2008; Faulwetter et al., 2009), while the plant community advances transported oxygen into the depth of the wetlands, provides root surfaces for rhizosphere reactions, and vents gases to the atmosphere. The plant processes are significantly affected by plant community composition and weather conditions (Towler et al., 2004; Stein and Hook, 2005; Stein et al., 2006; Zhu et al., 2007; Wang et al., 2008; Taylor et al., 2010).

Surface flow wetlands have a much more direct interchange with the atmosphere for the supply of oxygen and nitrogen, as well as the emissions of GHGs. They can be variable in shape and are generally less than 0.5 meters deep. Surface wetlands minimize clogging problems, but they can have significant loss of treatment as a result of channel flow. There are reasonably functional models for wetland design optimized for either carbon or nitrogen removal (Stone et al., 2002; Stone et al., 2004; Stein et al., 2006; Stein et al., 2007a). The management of GHGs (principally CH_4 and N_2O) from treatment wetlands is somewhat similar to managing GHGs in rice (Freeman et al., 1997; Tanner et al., 1997; Fey et al., 1999; Johansson et al., 2003; Mander et al., 2005a; Mander et al., 2005b; Teiter and Mander, 2005; Picek et al., 2007; Maltais-Landry et al., 2009; Wu et al., 2009).

Of particular importance is the maintenance of wetland oxidative/reductive potential conditions sufficiently positive to avoid CH_4 production (Tanner et al., 1997; Insam and Wett, 2008; Seo and DeLaune, 2010). This requires higher levels of oxygen and lower levels of available carbon. It has been reported that the fluxes of N_2O and CH_4 from treatment wetlands are generally below 10 mg N_2O-N m^{-2} d^{-1} and 300 mg CH_4-C m^{-2} d^{-1} (Mander et al., 2005a; Søvik et al., 2006). The management of N_2O emissions is complicated by the fact that nitrates are often present in the wastewaters or drainage waters. This nitrate will be denitrified under the prevailing anaerobic condition of the treatment wetlands—it is one of treatment wetland's critical functions. However, it is important that the preponderance of denitrification proceeds to completion, with the ultimate production of inert di-nitrogen gas. Complete denitrification requires higher carbon/nitrogen ratios

(Klemedtsson et al., 2005; Hwang et al., 2006; Hunt et al., 2007). Thus, there is an important balance between sufficient carbon for complete denitrification and copious carbon that can drive wetlands into the low reduction/oxidation conditions associated with CH_4 production.

Estimation methods are very complicated and case-based. In an approximate estimation manner that considers wetlands very similar to cropland, treatment wetlands of animal manure are GHG sinks more than sources. The CH_4 and N_2O emission from wetland treatment of animal manure could be negligible. The critical activity data include hydraulic load; inflow water composition, especially carbon and nitrogen; pretreatments such as solids removal or nitrification; amendments; and drying cycles. Critical ancillary data include rainfall, temperature, wind speed, storm events, changes in livestock stocking rates, cropping/tillage systems, and fertilization timing/rates.

5.4.11 Thermo-Chemical Conversion

Method for Estimating Emissions from Solid Manure Storage and Treatment – Thermochemical Conversion

- No method is provided as CH_4 and N_2O emissions are considered negligible.

Combustion, the most primitive and exothermic form of thermochemical treatment of livestock waste, has been in use since antiquity; however, its use for large-scale livestock waste treatment has generally been hampered by economic, health, and environmental quality issues (Florin et al., 2009). Principal among these issues has been components that degrade air quality, including GHGs (mainly CO_2). Nonetheless, thermochemical treatment of livestock manure has attributes that continue to attract efforts to make it economically and environmentally effective (Raman et al., 1980; He et al., 2000; He et al., 2001; Ocfemia et al., 2006; Ro et al., 2007; Cantrell et al., 2008a; Cantrell et al., 2008b; Powlson et al., 2008; Cantrell et al., 2009; Dong et al., 2009; Jin et al., 2009; Ro et al., 2009; Xiu et al., 2009; Cantrell et al., 2010a; Cantrell et al., 2010b; Stone et al., 2010; Wang et al., 2011; Xiu et al., 2011).

Recently, pyrolysis/gasification has received much interest for its treatment of livestock waste. There have also been advances in the cleaning of exhaust gases (He et al., 2001; Ro et al., 2007; Cantrell et al., 2008a; Dong et al., 2009; Xiu et al., 2009; Xiu et al., 2011). Pyrolysis/gasification offers three principal end products: syngas, bio-oil, and biochar (Cantrell et al., 2008a; Xiu et al., 2011). The quality and quantity of end products will vary with feedstock, exposure time, and pyrolysis/gasification temperature. The syngas can be used for direct combustion or to run an electrical generator (Ro et al., 2010). It can also be used via Fischer-Tropsch conversion for production of liquid fuel (Cantrell et al., 2008a). Pyrolysis/gasification for syngas and eventual liquid fuel production is a very attractive potential business model for specific agricultural fuels.

In terms of GHG emission, treatment of flue gas from combustion and utilization of syngas from pyrolysis/gasification are critical. The thermal processes with a flue gas clean-up unit and syngas utilization unit should minimize the GHG emission from the thermal conversion processes.

In order to estimate the daily emissions of CH_4 and N_2O the following information is needed: type of thermal conversion processes; detailed information on the process, such as with/without flue gas clean-up unit or syngas utilization unit; inflow composition, such as moisture, carbon, and nitrogen; and mass flow through the process, including mass in, flue gas/syngas, and ash/biochar. The measurements can be based on dietary changes or seasonal timeframe, which is decided by individual farm entity. However, due to the dynamic nature of manure piles and the rapid changes that can occur in chemical and physical composition, frequent measurements are recommended to ensure accuracy of the estimation. The total energy balance of the system should also be known. For instance, the carbon credits of biochar cannot be claimed while ignoring the energy required to create the biochar. The effectiveness of the exhaust gas cleaning process in removing air quality degrading components must be certified.

Due to the nature of thermal conversion, much lower emissions (CH_4 and N_2O,) are generated from the thermal conversion compared with other storage or treatment methods. The CH_4 and N_2O emissions from complete thermal conversion processes are relatively small and negligible.

5.4.12 Limitations and Uncertainty in Manure Management Emissions Estimates

For temporary and long-term storage, composting, and aerobic lagoons, the IPCC Tier 2 methodology is used to estimate CH_4 emissions. The maximum CH_4 production capabilities (B_0) for ruminant animals are U.S. specific values from the U.S. EPA Inventory of U.S. GHG Emissions and Sinks. IPCC estimates that the uncertainty associated with these country-specific factors is ±20 percent. B_0 values for other animal values are IPCC defaults and have an associated uncertainty of ±30 percent. The MCFs provided in the Guidelines for solid, slurry, and solid/slurry manure are from the IPCC Guidance and have an estimated uncertainty of ±30 percent. The B_0 and MCF values provided are intended for use at the national level, thus application of these factors at the entity level may result in higher uncertainty.

A modified Tier 2 approach is provided for estimating CH_4 emissions from anaerobic digesters. The leak rates for different digester types is taken from the Clean Development Mechanism's methodological tool for project and leakage emissions from anaerobic digesters (CDM, 2012). The Clean Development Mechanism's leak rates are based on IPCC (2006), Flesch et al. (2011), and Kurup (2003). The leakage rate taken from Flesch et al. (2011) is based on measurements taken from an Integrated Manure Utilization System installed in Alberta, Canada. The system processes 100 metric tons of manure daily and was the most technologically advanced system available at the time of the study. The studies performed by Kurup (2003) were based on a system located in Kerala, India. No uncertainty estimates are provided for these leak rates; however, the actual leak rate of an entity may differ due to differences in technology, maintenance, or other factors.

The Sommer model (Sommer et al., 2004) is recommended for estimating CH_4 emissions from anaerobic lagoons, runoff holding ponds, and storage tanks. Similar to the IPCC Tier 2 methods used for stockpiles, composting, and aerobic lagoons, the Sommer model requires B_0 values from IPCC. The degradable and nondegradable volatile solids can be calculated using the B_0 and potential CH_4 yield or a default value from Møller at al. (2004). The default values presented are based on typical concentrations on Danish cattle and pig slurries; values do not differentiate between type of cattle or diet of the animal and thus there is higher relative uncertainty associated with using the default values.

Sommer et al. (2004) performed an analysis to determine the sensitivity of emission estimates towards different factors. One factor considered is the effect of slurry storage temperature on CH_4 emissions. Sommer et al. (2004) applied average monthly temperatures for seven different locations (all Nordic countries) at constant volatile solids and management. When compared to the model results for Denmark (which are calibrated to correspond with IPCC methodology), the emissions estimates varied from -1 to +36 percent for pig slurry and -23 to +1 percent for cattle slurry. Given that the climatic conditions of the United States differs from Nordic countries, the variation as a result of slurry storage temperature is expected to be greater.

IPCC methodology or modified methodology is used to estimate the N_2O emissions from temporary stack and long-term storage, composting, and aerobic lagoons. IPCC reports large uncertainties with the default emission factors applied (-50 percent to +100 percent). These emission factors were intended for use at the national level and do not take into account varying temperature, moisture content, aeration, manure nitrogen content, metabolizable carbon, duration of storage, and other aspects of treatment for different entities, thus the uncertainty is expected to be higher than reported by IPCC.

The methods recommend that the user send manure samples to a laboratory to obtain an estimate of the volatile solids, NH_3, and nitrogen content of manure. A measurement of manure characteristics can help minimize uncertainty by providing an entity-specific value that takes into account animal and diet characteristics. If laboratory-tested volatile solids values are not available, default values from the American Society of Agricultural and Biological Engineers (ASABE) can be applied. ASABE provides default manure characteristics based on data from published and unpublished information. These values are arithmetic averages and may not represent the differences in animal age, diet, usage, productivity, and management. There is a higher amount of uncertainty associated with the use of ASABE values but there is no quantified uncertainty provided for these values. Note that within the standard cited below there are equations provided that allow for farm-specific values to be determined based on animal characteristics and diet composition. The table below is intended to provide 'average' values, but where farm data are available, equations should be used in order to provide more estimates that better reflect farm conditions and practices.

Available default values and uncertainty information is included in Table 5-32.

Table 5-32: Available Uncertainty Data for Emissions from Manure Management

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total Dry Manure – Beef Finishing Cattle		kg dry manure/animal/ day	2.4	-20	20			ASABE (2005)
Total Dry Manure – Beef Cow (confinement)		kg dry manure/animal/ day	6.6	-20	20			ASABE (2005)
Total Dry Manure – Beef Growing calf (confinement)		kg dry manure/animal/ day	2.7	-20	20			ASABE (2005)
Total Dry Manure – Dairy Lactating cow		kg dry manure/animal/ day	8.9	-20	20	8.7	11.3	ASABE (2005)
Total Dry Manure – Dairy Dry cow		kg dry manure/animal/ day	4.9	-20	20	8.8	11.2	ASABE (2005)
Total Dry Manure – Dairy Heifer		kg dry manure/animal/ day	3.7	-20	20			ASABE (2005)
Total Dry Manure – Dairy Veal 118 kg		kg dry manure/animal/ day	0.12	-20	20			ASABE (2005)
Total Dry Manure – Horse Sedentary 500 kg		kg dry manure/animal/ day	3.8	-20	20			ASABE (2005)
Total Dry Manure – Horse Intense exercise 500 kg		kg dry manure/animal/ day	3.9	-20	20			ASABE (2005)
Total Dry Manure – Poultry Broiler		kg dry manure/animal/ day	0.03	-20	20			ASABE (2005)
Total Dry Manure – Poultry Turkey (male)		kg dry manure/animal/ day	0.07	-20	20			ASABE (2005)
Total Dry Manure – Poultry Turkey (females)		kg dry manure/animal/ day	0.04	-20	20			ASABE (2005)
Total Dry Manure – Poultry Duck		kg dry manure/animal/ day	0.04	-20	20			ASABE (2005)
Total Dry Manure – Layer		kg dry manure/animal/ day	0.02	-20	20			ASABE (2005)
Total Dry Manure – Swine Nursery pig (12.5 kg)		kg dry manure/animal/ day	0.13	-20	20			ASABE (2005)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total Dry Manure – Swine Grow finish (70 kg)		kg dry manure/animal/ day	0.47	-20	20			ASABE (2005)
Total Dry Manure – Swine gestating sow 200 kg		kg dry manure/animal/ day	0.5	-20	20			ASABE (2005)
Total Dry Manure – Swine Lactating sow 192 kg		kg dry manure/animal/ day	1.2	-20	20			ASABE (2005)
Total Dry Manure – Swine Boar 200 kg		kg dry manure/animal/ day	0.38	-20	20			ASABE (2005)
Volatile solids – Beef Finishing cattle	VS	kg VS/kg dry manure	0.81	-25	25			ASABE (2005)
Volatile solids – Beef Cow (confinement)	VS	kg VS/kg dry manure	0.89	-25	25			ASABE (2005)
Volatile solids – Beef Growing calf (confinement)	VS	kg VS/kg dry manure	0.85	-25	25			ASABE (2005)
Volatile solids – Dairy Lactating cow	VS	kg VS/kg dry manure	0.84	-25	25			ASABE (2005)
Volatile solids – Dairy Dry cow	VS	kg VS/kg dry manure	0.85	-25	25			ASABE (2005)
Volatile solids – Dairy Heifer	VS	kg VS/kg dry manure	0.86	-25	25			ASABE (2005)
Volatile solids – Dairy Veal 118 kg	VS	kg VS/kg dry manure		-25	25			ASABE (2005)
Volatile solids – Horse Sedentary 500 kg	VS	kg VS/kg dry manure	0.79	-25	25			ASABE (2005)
Volatile solids – Horse Intense exercise 500 kg	VS	kg VS/kg dry manure	0.79	-25	25			ASABE (2005)
Volatile solids – Poultry Broiler	VS	kg VS/kg dry manure	0.73	-25	25			ASABE (2005)
Volatile solids – Poultry Turkey (male)	VS	kg VS/kg dry manure	0.8	-25	25			ASABE (2005)
Volatile solids – Poultry Turkey (females)	VS	kg VS/kg dry manure	0.79	-25	25			ASABE (2005)
Volatile solids – Poultry Duck	VS	kg VS/kg dry manure	0.58	-25	25			ASABE (2005)
Volatile solids – Layer	VS	kg VS/kg dry manure	0.73	-25	25			ASABE (2005)
Volatile solids – Swine Nursery pig (12.5 kg)	VS	kg VS/kg dry manure	0.83	-25	25			ASABE (2005)
Volatile solids – Swine Grow finish (70 kg)	VS	kg VS/kg dry manure	0.8	-25	25			ASABE (2005)
Volatile solids – Swine gestating sow 200 kg	VS	kg VS/kg dry manure	0.9	-25	25			ASABE (2005)
Volatile solids – Swine Lactating sow 192 kg	VS	kg VS/kg dry manure	0.83	-25	25			ASABE (2005)
Volatile solids – Swine Boar 200 kg	VS	kg VS/kg dry manure	0.89	-25	25			ASABE (2005)
Total nitrogen at a given day – beef finishing cattle		kg N/kg dry manure	0.07					ASABE (2005)
Total nitrogen at a given day – beef cow (confinement)		kg N/kg dry manure	0.03					ASABE (2005)
Total nitrogen at a given day – beef growing calf (confinement)		kg N/kg dry manure	0.05					ASABE (2005)
Total nitrogen at a given day – dairy lactating cow		kg N/kg dry manure	0.05					ASABE (2005)
Total nitrogen at a given day – dairy dry cow		kg N/kg dry manure	0.05					ASABE (2005)
Total nitrogen at a given day – dairy heifer		kg N/kg dry manure	0.03					ASABE (2005)
Total nitrogen at a given day – dairy veal 118 kg		kg N/kg dry manure	0.13					ASABE (2005)
Total nitrogen at a given day – Horse Sedentary 500 kg		kg N/kg dry manure	0.02					ASABE (2005)
Total nitrogen at a given day – Horse Intense Exercise		kg N/kg dry manure	0.04					ASABE (2005)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total nitrogen at a given day – poultry, broiler		kg N/kg dry manure	0.04					ASABE (2005)
Total nitrogen at a given day – poultry, turkey (male)		kg N/kg dry manure	0.06					ASABE (2005)
Total nitrogen at a given day – poultry, turkey (females)		kg N/kg dry manure	0.06					ASABE (2005)
Total nitrogen at a given day – poultry, duck		kg N/kg dry manure	0.04					ASABE (2005)
Total nitrogen at a given day – layer		kg N/kg dry manure	0.07					ASABE (2005)
Total nitrogen at a given day – swine nursery pig (12.5 kg)		kg N/kg dry manure	0.09					ASABE (2005)
Total nitrogen at a given day – swine grow finish (70 kg)		kg N/kg dry manure	0.08					ASABE (2005)
Total nitrogen at a given day – swine gestating sow 200 kg		kg N/kg dry manure	0.06					ASABE (2005)
Total nitrogen at a given day – swine lactating sow 192 kg		kg N/kg dry manure	0.07					ASABE (2005)
Total nitrogen at a given day – swine boar 200 kg		kg N/kg dry manure	0.07					ASABE (2005)
Methane Conversion Factor (MCF) [a]– Dairy Cow	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Cattle	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Buffalo	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Market Swine	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Breeding Swine	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Layer (Dry)	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Broiler	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Turkey	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Duck	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Sheep	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Goat	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Horse	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Mule/Ass	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Buffalo	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – In vessel manure composting	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Static pile manure composting	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Intensive windrow	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Passive windrow	MCF	%		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Beef Replacement Heifers	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Dairy Replacement	B_o	m^3 CH_4/kg VS	0.17	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Mature Beef Cows	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Steers (>500 lbs)	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Maximum Methane Producing Capacities – Stockers (All)	B_o	m^3 CH_4/kg VS	0.17	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Cattle on Feed	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Dairy Cow	B_o	m^3 CH_4/kg VS	0.24	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Cattle	B_o	m^3 CH_4/kg VS	0.19	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Buffalo[b]	B_o	m^3 CH_4/kg VS	0.1					IPCC (2006)
Maximum Methane Producing Capacities – Market Swine	B_o	m^3 CH_4/kg VS	0.48	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Breeding Swine	B_o	m^3 CH_4/kg VS	0.48	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Layer (dry)	B_o	m^3 CH_4/kg VS	0.39	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Layer (wet)	B_o	m^3 CH_4/kg VS	0.39	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Broiler	B_o	m^3 CH_4/kg VS	0.36	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Turkey	B_o	m^3 CH_4/kg VS	0.36	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Duck	B_o	m^3 CH_4/kg VS	0.36	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Sheep	B_o	m^3 CH_4/kg VS	0.19	-20	20			IPCC (2006)
Maximum Methane Producing Capacities – Feedlot sheep	B_o	m^3 CH_4/kg VS	0.36	-20	20			IPCC (2006)
Maximum Methane Producing Capacities – Goat	B_o	m^3 CH_4/kg VS	0.17	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Horse	B_o	m^3 CH_4/kg VS	0.3	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Mule/Ass	B_o	m^3 CH_4/kg VS	0.33	-30	30			IPCC (2006)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Digesters with steel or lined concrete or fiberglass digesters with a gas holding system (egg shaped digesters) and monolithic construction	$EF_{CH4,\ leakage}$	%	2.8					CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – UASB type digesters with floating gas holders and no external water seal	$EF_{CH4,\ leakage}$	%	5					CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Digesters with unlined concrete/ferrocement/brick masonry arched type gas holding section; monolithic fixed dome digesters	$EF_{CH4,\ leakage}$	%	10					CDM (2012)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Other digester configurations	$EF_{CH4, leakage}$	%	10					CDM (2012)
Temporary storage of liquid/slurry manure –N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.005	-50	100			U.S. EPA (2011)
Long-term storage of solid manure – N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.002	-50	100			U.S. EPA (2011)
Long-term storage of slurry manure – N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.005	-50	100			U.S. EPA (2011)
Cattle and Swine Deep Bedding (Active Mix)- N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.07					IPCC (2006)
Cattle and Swine Deep Bedding (No Mix)- N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.01					IPCC (2006)
Pit Storage Below Animal Confinements- N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.002					IPCC (2006)
Natural aeration aerobic lagoons – N_2O conversion factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.01	-50	100			IPCC (2006)
Forced aeration aerobic lagoons – N_2O conversion factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.005	-50	100			IPCC (2006)
N_2O emission factor for liquid storage – uncovered liquid manure with a crust[c]	EF_{N2O}	kg N_2O-N/kg N	0.8	-50	100			IPCC (2006)
N_2O emission factor for liquid storage – uncovered liquid manure without a crust[c]	EF_{N2O}	kg N_2O-N/kg N	0	-50	100			IPCC (2006)
N_2O emission factor for liquid storage – covered liquid manure[c]	EF_{N2O}	kg N_2O-N/kg N	0	-50	100			IPCC (2006)
Manure Management – Multiple Sources – collection efficiency, covered storage (with or without crust)	η	Percentage	1					Sommer et al. (2004)
Manure Management – Multiple Sources – collection efficiency, uncovered storage with crust formation	η	Percentage	0					Sommer et al. (2004)
Manure Management – Multiple Sources – collection efficiency, uncovered storage without crust formation	η	Percentage	-0.40					Sommer et al. (2004)
Manure Management – Multiple Sources – Rate correcting factors (b_1)	b_1	Dimensionless	1					Sommer et al. (2004)
Manure Management – Multiple Sources – Rate correcting factors (b_2)	b_2	Dimensionless	0.01					Sommer et al. (2004)
Manure Management – Multiple Sources – Arrhenius parameter, cattle	A	g CH_4 /kg VS/hr	43.33					Sommer et al. (2004)
Manure Management – Multiple Sources – Arrhenius parameter, swine	A	g CH_4 /kg VS/hr	43.21					Sommer et al. (2004)
Potential methane yield of the manure cattle	$E_{CH4, pot}$ -	kg CH_4/kg VS	0.48					Sommer et al. (2004)
Potential methane yield of the manure - swine	$E_{CH4, pot}$	kg CH_4/kg VS	0.5					Sommer et al. (2004)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Temporary stack and long-term stockpile – Ratio degradable volatile solids to total volatile solids - cattle liquid manure	VS_d/VS_T	Unitless	0.46					Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio degradable volatile solids to total volatile solids - swine liquid manure	VS_d/VS_T	Unitless	0.89					Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio Non-degradable volatile solids to total volatile solids - cattle liquid manure	VS_{nd}/VS_T	Unitless	0.54					Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio non-degradable volatile solids to total volatile solids – swine liquid manure	VS_{nd}/VS_T	Unitless	0.11					Møller et al. (2004)

[a] The values for methane conversion factor (MCF) vary depending on the temperature and the manure management system. IPCC (2006) provides estimated uncertainty ranges for these MCFs.
[b] There are no data for North America region; the data from Western Europe are used to calculate the estimation. There is no reported uncertainty for this adapted value.
[c] IPCC (2006) reports large uncertainties with default N_2O emission factors. The N_2O EF values vary depending on the animal species and temperature of the manure management system.

5.5 Research Gaps

Research gaps have been identified for animal production systems, covering activity data, as well as key areas that would facilitate more accurate estimation of emissions from enteric fermentation and manure management systems. Recommendations are discussed below.

5.5.1 Enteric Fermentation

Cattle

Future research related to improving emissions estimates should be aimed at expanding the options within existing models to better describe an individual farm system and incorporate more options for mitigation strategies to see how emissions might change with implementation of these strategies as well as consider the interactive effects of multiple strategies.

Beef Cow-Calf, Bulls, Stocker, and Sheep

Key data needs include measurement/prediction of feed intake on pasture, measurement/prediction of CH_4 from grazing animals (larger numbers of animals), and methods by which to characterize range forage and intake under production conditions.

Feedlot

There is a need for equations and models to accurately predict enteric CH_4 emissions from cattle and sheep fed high-concentrate finishing diets.

Dairy

One of the largest research gaps is the lack of basic data related to emissions from calves, heifers, and dry cow housing systems. In addition, there is a need for equally consistent and reliable methods for measuring relative differences in emissions associated with the implementation of a variety of management practices. Further model development for estimating emissions should include an expansion of options to describe the production facility and inclusion of management practices that can be adopted to mitigate emissions.

Swine

Future research related to improving emissions estimates should be aimed at expanding the options within these models to better describe an individual farm system and incorporate more options for management and mitigation strategies to see how emissions might change with implementation of different practices. Minimally, the diet considerations in Holos need to be incorporated into the MANURE model and expanded to reflect production phase.

Poultry

Future research related to improving emissions estimates should be aimed at expanding the options within these models to better describe an individual farm system and incorporate more options for management and mitigation strategies to see how emissions might change with implementation of different practices.

5.5.2 Manure Management

Greenhouse gas emissions from a variety of manure management systems have been developed from a limited number of studies and a limited number of potential variations in management and the environmental conditions around a particular manure management system. The largest deficiency in the current GHG studies is the lack of characterization of the temporal variation in the GHG emissions from different systems and the spatial variation in GHG emissions induced by meteorological conditions among specific locations. In general, the research needed to develop a more complete understanding of the GHG emissions can be summarized as:

- Develop data bases from research observations of commercial facilities that characterize the storage system, time in storage, environmental conditions and location, and the attributes of the manure source, e.g., type of animal, diet, loading rate.
- Utilize the databases to derive simulation models to quantify the GHG emissions from different manure management systems.
- Validate the models using independent observations from manure management systems distributed around the United States.
- Develop operational models capable of being applied to production scale systems which utilize simple parameters as input variables and produce results in agreement with the more complex simulation models.
- Utilize these models to develop potential strategies which could be employed to mitigate GHG emissions from manure management systems.

Temporary Stack and Long-Term Stockpile

Methane emission data from solid storages in different regions under different climates are limited. In order to develop a more accurate model to estimate the CH_4 emission from solid manure storages, in-depth studies are needed to integrate temperature, storage time, storage method, and mass flow with CH_4 emission in different regions. As for N_2O emission, systematically collecting more intense data (a variety of spatial and temporal scales) from different regions will be a good first step toward accurate N_2O emission models. Once these data are collected and used to develop/validate models, work will likely be needed to develop farmer-friendly models using simple farm parameters as input variables, resulting in emissions estimates that are correlated with those of more complex models. For example, these models, if synchronized, could form part of a comprehensive manure stewardship toolkit.

There is a paucity of data on CH_4 and N_2O emissions from open lot (beef feedlots and dairies) pen surfaces and runoff control structures and on the chemical and physical factors controlling those emissions.

Composting

Greenhouse gas emissions data from composting in different regions under different operational conditions are limited. A good first step toward an accurate GHG emissions model would be to collect more data from different regions and different operational conditions. Consequently, in-depth studies integrating compost pile size/surface area, pile shape, aeration rate, storage time, composting temperature, etc., with GHG emissions need to be conducted to develop complex models describing GHG emissions from composting. Furthermore, work will likely be needed to develop farmer-friendly models using simple farm parameters as input variables, resulting in emission estimations that are correlated with those of more complex models.

There have been some studies performed to estimate the emission factors for N_2O from composting manure in different systems and for different livestock categories. (Fukummoto et al., 2003; Szanto et al., 2006) have conducted studies on composting swine manure at specific ambient temperatures. Factors have been presented in the studies but there is significant uncertainty due to the limited data available. Further research is needed to refine these emission factors as well as develop factors for other animals.

Aerobic Lagoon

In-depth studies are needed to integrate lagoon depths, aeration rate, pH, temperature, and nutrient conditions of manure with GHG emissions, which will facilitate the development of comprehensive models to predict GHG emissions under different operational and climate conditions. Simplified and farm-friendly models using farm operational parameters as inputs should be developed to help farms estimate the GHG emissions at the entity level.

Anaerobic Lagoon, Runoff Holding Pond, and Storage Tanks

All models to estimate GHG emissions from liquid manure storage are relatively inaccurate, due to the complexity and variety of livestock manure operations. In order to develop a more accurate model to estimate emissions from liquid manure storages, in-depth studies are needed to integrate manure storage configuration, temperature, storage time, storage method, mass flow, and surface turbulence with emissions in different regions. In addition, systematically collecting more data from different regions will be very helpful to develop more statistically accurate models to estimate GHG emissions.

Anaerobic Digestion

Changes in chemical oxygen demand, volatile solids, total solids, and nitrogen in the anaerobic digestion process are indirectly linked to GHG emissions from post-treatment of anaerobic digestion effluent. The effectiveness of anaerobic digestion at mitigating GHG emissions has been studied intensively. However, anaerobic digestion effluent can lead to GHG emissions. More in-depth studies are needed to develop integrated models that can accurately predict the overall GHG emission from the combination of anaerobic digestion and post-treatment approaches.

Combined Aerobic Treatment Systems

Methods and techniques to reduce the capital and operating costs are needed. There is also a need to develop better ways to conserve and derive energy from the waste material. There is a paucity of data on GHG emissions from these systems and development of emission models will require integration of data characterizing these systems and the climatic conditions in order to develop these models. These models will need to be validated against observed data.

Nutrient Removal

Various methods of nitrogen removal, such as biological nitrogen removal, Anamox, NH_3 stripping, ion exchange, and struvite crystallization, should be investigated at commercial-scale animal operations under different climate conditions. Characteristics of manure, mass flow, and gas emissions should be closely monitored in order to provide the data needed to construct relatively precise estimation models. In addition, further research is needed to pilot innovative beef and dairy GHG emission reduction strategies in feedlots and dairies.

Constructed Wetland

Although there are numerous papers published about various aspects of treatment wetland effectiveness and emissions, there currently is not an established method for calculation of GHG emissions from any of the treatment wetland types. Moreover, there are not sufficient unifying publications to suggest that a reliable method could be established within the scope of this report. A more robust and extensive database on GHG emissions from treatment wetlands is needed. Concomitantly, there is a need for better predictive equation and models.

Thermo-Chemical Conversion

More studies are needed on the effects of thermal conversion of animal manure on GHG emission in order to conclude detailed emission profiles corresponding to different type of manure. These studies would entail detailed observations of the manure conversion system along with GHG emissions and information on the environmental conditions.

Appendix 5-A: Enteric CH_4 from Feedlot Cattle – Methane Conversion Factor (Ym)

As noted in the Beef Production Systems section (Section 5.3.2.2), a modified IPCC (2006) method is proposed to estimate enteric CH_4 emissions from finishing beef cattle. For this report, a baseline scenario based on typical U.S. beef cattle feeding conditions was established and baseline values were set based on published research. To estimate methane emissions, emission values are modified using adjustment factors that are based on changes in animal management and feeding conditions from the baseline scenario. This appendix presents background information on the baseline scenario and adjustment factors.

The following baseline scenarios are established for beef cattle in U.S. feedlots:

1. Medium to large frame steer (or heifer) yearlings are fed a high concentrate finishing diet containing <=10 percent forage in diet dry matter (= to 8 to 18 percent NDF) in dry-lot, soil-surfaced pens.
2. The grain portion of the diet is at least 70 percent of diet dry matter.
3. The grain source is steam flaked (SFC) or high moisture corn (HMC).
4. The dietary crude protein concentration is 12.5 to 13.5 percent of diet dry matter (Vasconcelos and Galyean, 2007).
5. The dietary ruminally degradable protein (DIP or RDP) concentration is 7.5 to 9 percent of diet dry matter (Vasconcelos and Galyean, 2007).
6. The diet contains monensin (Rumensin, Elanco Animal Health) at recommended concentrations (Vasconcelos and Galyean, 2007).
7. Diets for heifers contain melengestrol acetate (MGA) at the recommended concentrations (Vasconcelos and Galyean, 2007).
8. Cattle are implanted with an estrogenic implant throughout the feeding period (Vasconcelos and Galyean, 2007).
9. No beta-agonist is fed.
10. The diet contains no supplemental fat (vegetable oil, yellow grease, etc.) and has a total fat concentration of less than 4.5 percent of diet dry matter.
11. Enteric CH_4 emission is three percent of gross energy intake (GEI: (IPCC, 2006).
12. The dietary forage is chopped alfalfa, sorghum, or grass hay at seven to 10 percent of diet dry matter.
13. The diet contains minerals and vitamins at the recommended level (NRC, 2000).
14. Temperatures are mild/moderate during the feeding period.
15. Cattle are slaughtered at an average body weight of approximately 582 kg (1,280 lb.) (KSU, 2012).
16. Average dressing percent is 61 percent.
17. Cattle are fed 150 days.

The Ym adjustment factors for feedlot cattle fed high-concentrate diets in Table 5-11 were determined based on the following literature reviews and analyses.

Ionophores: On average, the feeding of ionophores decreases DMI by about five percent (Delfino et al., 1988; Vogel, 1995; Robinson and Okine, 2001; Tedeschi et al., 2003) and decreases ADG by about two percent (Delfino et al., 1988; Tedeschi et al., 2003). Feeding ionophores decreases enteric methane emissions approximately 20 percent for the first two to four weeks on feed (Tedeschi et al., 2003; Guan et al., 2006). Therefore, over a 150-day feeding period, overall enteric methane emissions are decreased approximately 4 percent. Because of an increase in the gain:feed ratio, enteric methane emissions per unit of production are decreased when ionophores are fed.

Supplemental Fat: For each one percent increase in supplemental fat (up to a maximum of four percent added fat), enteric methane emissions (as a percentage of gross energy intake) decrease approximately 3.8 to 5.6 percent (Zinn and Shen, 1996; Beauchemin et al., 2008; Martin et al., 2010). A conservative value of four percent per one percent increase in supplemental fat is recommended because many fat sources used in the industry are partially saturated and may have less effect on enteric CH_4 production than the highly unsaturated fats used in most studies. For example if three percent supplemental fat is added to the diet, then CH_4 production is decreased 12 percent (three percent added fat times four percent is equivalent to a 12 percent decrease). The revised enteric CH_4 emission is 2.64 percent of GEI (three percent baseline * 0.88 = 2.64 percent of GEI). Many distiller's grains contain approximately 8 to 12 percent fat. Addition of distiller's grain may serve as a source of supplemental fat, and thus decrease enteric CH_4 (McGinn et al., 2009). However Hales et al. (2013) noted that feeding increasing concentrations of Wet Distillers Grains with Solubles (WDGS) in equal-fat diets increased enteric CH_4, likely due to the increased NDF intake.[12]

Grain processing & Grain source: Grain processing directly affects enteric CH_4 production via its effects on ruminal fermentation. Enteric CH_4 emissions, as a percent of GEI, are 20 percent greater with diets based on DRC than in diets based on steam-flaked corn (SFC) or high moisture corn (HMC) (Archibeque et al., 2006; Hales et al., 2012). More extensive grain processing may also improve the gain:feed ratio about 10 percent (Owens et al., 1997; Zinn and Barajas, 1997) and may, decrease manure CH_4 emissions via decreased fecal starch excretion (Zinn and Barajas, 1997; Hales et al., 2012). Enteric CH_4 emissions are 20 to 40 percent greater with finishing diets based on barley than diets based on corn; presumably because of the lower starch and higher fiber content of barley (Benchaar et al., 2001; Beauchemin and McGinn, 2005). A mean (30%) for these studies is recommended for a barley adjustment factor.

Dietary Forage and Grain Concentration effects: Limited data exists to evaluate effects of dietary forage and grain concentration on enteric methane production from beef cattle that are fed typical U.S-based, high concentrate finishing diets. Equations from Ellis et al. (2007; 2009) illustrate the effects of dietary forage, NDF, and starch on enteric CH_4 production. In particular, the following 10 equations illustrate the relationships:

- CH_4 (MJ/day) = 3.96 + 0.561 × DMI (kg/day)
- CH_4 (MJ/day) = 4.79 + 0.0492 × Forage (%)
- CH_4 (MJ/day) = 5.58 + 0.848 × NDF (kg/day)
- CH_4 (MJ/day) = 5.70 + 1.41 × ADF (kg/day)
- CH_4 (MJ/day) = 2.29 + 0.670 × DMI (kg/day)
- CH_4 (MJ/day) = 4.72 + 1.13 × Starch (kg/day)
- CH_4 (MJ/day) = -1.01 + 2.76 × NDF (kg/day) +0.722 × Starch (kg/day)
- CH_4 (MJ/day) = 2.68 – 1.14 (Starch:NDF) + 0.786 × DMI (kg/day)
- CH_4 (MJ/day) = 2.50 = 0.367 × Starch (kg/day) + 0.766 × DMI (kg/day)
- CH_4 (MJ/day) = 2.70 + (1.16 × DMI (kg/day)) – (15.8 × ether extract (kg/day))

[12] Ym is adjusted for distiller grains by changes in fat content and grain concentration. For example, a 30 percent concentration of distiller grains in the finishing diet will typically increase the dietary fat level by 2 to 3 percent and decrease the grain content by 25 to 30 percent. The resulting change in Ym is a decrease by 8 percent to account for increase in fat content and an increase of 10 percent to account for a decrease in grain content (i.e., Ym = 3% x 0.92 x 1.10 = 3.036 %).

To develop adjustment factors for grain concentrations in diets, artificial data sets were created that varied in forage (range of 5 to 25 percent), NDF (range 10 to 20 percent), fat (range of 3 to 6 percent), and starch (range of 30 to 60 percent of diet dry matter) content. Using these data sets, enteric CH_4 emissions were estimated using the appropriate equation(s) of Ellis et al. (2007; 2009). Effects of dietary changes on enteric CH_4 were then determined by linear regression analysis. On average, enteric CH_4 production (MJ/day) increased five percent for each one percent increase in dietary forage concentration; increased 13 percent for each one kg increase in dietary NDF intake, increased five percent for each one kg increase in starch intake and decreased five percent for each one unit increase in the dietary starch:NDF ratio. Small increases in forage concentration from the baseline value had small effects on Ym; whereas, greater increases had a larger effect (Hales et al., 2012; Hales et al., 2014). An evaluation of these factors indicated an enteric CH_4 Ym adjustment factor of 10% for small increases in forage (and decreases in grain concentration) and a larger correction factor of 40 percent for greater changes (diet concentrate less than 45 percent). These factors are recommended for accounting for the grain concentration in finishing diets.

No Ym adjustment factor was explicitly modeled to account for the following dietary management factors:[13]

- Beta-agonists: Beta-agonists do not directly affect the Ym (i.e., enteric CH_4 emissions per unit of gross energy intake), therefore no adjustment factor is recommended. However, because of a 4 percent increase in feed efficiency, a 2.5 to 3.5% increase in hot carcass weight (HCW), and an increase in live body weight (Vasconcelos et al., 2008; Elam et al., 2009; Montgomery et al., 2009; Delmore et al., 2010; Radunz, 2011), enteric CH_4 emissions per unit of production are decreased when beta-agonists are fed.
- Melengestrol acetate (MGA: heifers only): Feeding MGA to heifers does not directly affect enteric CH_4 emissions. However, because of a nine percent increase in the gain:feed ratio (Hill et al., 1988; Kreikemeier and Mader, 2004) enteric CH_4 emissions per unit of production are decreased when MGA is fed.
- Direct Fed Microbials: Most direct fed microbials do not appear to directly affect enteric CH_4 emissions and effects on animal performance are somewhat variable (Krehbiel et al., 2003). No adjustment factor is recommended for the feeding of direct fed microbials.
- Dietary Crude Protein and Ruminal Degradable Protein (RDP): Dietary protein may potentially affect animal performance and enteric CH_4 emissions via effects on ruminal fermentation. However, there is no readily available data with modern feedlot diets with which to compare (Berger and Merchen, 1995; Robinson and Okine, 2001; Gleghorn et al., 2004; Cole et al., 2006; Wagner et al., 2010). There is no recommended Ym adjustment factor for dietary protein. Dietary protein may affect emissions of manure greenhouse gases (N_2O) and definitely affects NH_3 emissions (Todd et al., 2013).
- Implanting regimens: Implants do not directly affect enteric CH_4 emissions. However because of an increase in feed efficiency, live body weight, and HCW (Herschler et al., 1995; Robinson and Okine, 2001; Wileman et al., 2009), enteric CH_4 emissions per unit of production are decreased when implants are used.
- Ambient temperature: Cold and hot temperatures may potentially affect enteric CH_4 emission due to effects on feed intake, ruminal digestion and rate of passage (Young, 1981); however, the actual effects are not clear. Therefore no adjustment factor for environmental temperature is used. Cold temperatures may decrease CH_4, N_2O and NH_3 losses from pen

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[13] Although these management factors are not modeled to impact Ym, some of them do impact enteric CH_4 per unit of production. Hence, in evaluating methane intensity per unit of production, these factors would have an impact.

surfaces via effects on microbial activity in the manure. Conversely, warm temperatures may increase emissions from manure via increased microbial activity.

The IPCC Tier 2 model is currently the most useful for predicting emissions from cow-calf and stocker production, as well, as noted in the earlier cow-calf and stocker Sections (5.3.2.2). Enteric emissions from all cattle other than dairy cows and dairy heifers are estimated using the IPCC Tier 2 equation or the modified IPCC Tier 2 previously discussed for feedlot cattle. To use these equations, it is necessary to make sure the inputs to the equations are as accurate as possible. For DE (as a percentage of GE), we recommend using the feedstuffs composition table provided in NRC (1989) and Ewan (1989). Several feedstuffs from the table are included in Table 5-C-1. After review of the models, their strengths and limitations, models based on the Mills equations (e.g., DairyGEM, COWPOLL, IFSM) appear to be the most useful for predicting emissions from dairy cattle. The Mits3 equation recommended for calculating enteric CH_4 emissions from dairy cows and dairy heifers (used in DairyGEM/IFSM) requires different dietary input information than that required for the IPCC Tier II equation. Specifically, DairyGEM/IFSM requires the starch and ADF content of feeds. Because starch is nearly equivalent to NFC (which is starch + sugar + pectin) in high forage diets (dairy diets), we use NFC in the Mits3 equation (NFC = 100 – (NDF + CP + EE + Ash)). These values can be found in Appendix 5-B.

Appendix 5-B: Feedstuffs Composition Table

This table provided data inputs for enteric fermentation emissions calculations for cattle and sheep.

Table 5-B-1: Feedstuffs Composition Table (Preston 2013, except where noted for digestible energy)

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Alfalfa Cubes	x91	57	57	25	57		18	30	29	36	46	40	2.0	11	1.30	0.23	1.9	0.37	0.33	20
Alfalfa Dehydrated 17% CP	92	61	62	31	61	65.16	19	60	26	34	45	6	3.0	11	1.42	0.25	2.5	0.45	0.28	21
Alfalfa Fresh	24	61	62	31	61	62.54[b]	19	18	27	34	46	41	3.0	9	1.35	0.27	2.6	0.40	0.29	18
Alfalfa Hay Early Bloom	90	59	59	28	59	63.72	19	20	28	35	45	92	2.5	8	1.41	0.26	2.5	0.38	0.28	22
Alfalfa Hay Midbloom	89	58	58	26	58	61.79	17	23	30	36	47	92	2.3	9	1.40	0.24	2.0	0.38	0.27	24
Alfalfa Hay Full Bloom	88	54	54	20	54	55.71	16	25	34	40	52	92	2.0	8	1.20	0.23	1.7	0.37	0.25	23
Alfalfa Hay Mature	88	50	50	12	49	54.18	13	30	38	45	59	92	1.3	8	1.18	0.19	1.5	0.35	0.21	23
Alfalfa Seed Screenings	91	84	92	61	87		34		13	15			10.7	6	0.30	0.67				
Alfalfa Silage	30	55	55	21	55	60.71[c]	18	19	28	37	49	82	3.0	9	1.40	0.29	2.6	0.41	0.29	26
Alfalfa Silage Wilted	39	58	58	26	58	60.71[d]	18	22	28	37	49	82	3.0	9	1.40	0.29	2.6	0.41	0.29	26
Alfalfa Leaf Meal	89	60	60	30	60		26	15	16	24	34	35	3.0	10	2.88	0.34	2.2		0.32	39
Alfalfa Stems	89	47	47	7	46		11	44	44	51	68	100	1.3	6	0.90	0.18	2.5			
Almond Hulls	89	56	56	23	56	59.90	3	60	16	29	36	100	3.1	7	0.24	0.10	2.0	0.03	0.07	20
Ammonium Chloride	99	0	0	0	0		163	0	0	0	0	0	0.0		0.00	0.00	0.0	66.00	0.00	0
Ammonium Sulfate	99	0	0	0	0		132	0	0	0	0	0	0.0						24.15	
Apples	17	70	73	44	71		3	10	7	9	25	10	2.2	2	0.06	0.60	0.8			
Apple Pomace Wet	20	68	70	41	69		5	10	18	27	36	27	5.2	3	0.13	0.12	0.5		0.04	11
Apple Pomace Dried	89	67	69	40	68	56.69	5	15	18	28	38	29	5.2	3	0.13	0.12	0.5		0.04	11
Artichoke Tops (Jerusalem)	27	61	62	31	61		6		18	30	41	40	1.1	10	1.62	0.11	1.4			
Avocado Seed Meal	91	52	52	16	51		20		19	24			1.2	16						
Bahiagrass Hay	90	53	53	18	53	54.85	6	37	32	41	72	98	1.8	7	0.47	0.20	1.4		0.21	
Bakery Product Dried	90	90	100	68	94	81.31	11	30	3	9	30	0	11.5	4	0.16	0.27	0.4	2.25	0.15	33
Bananas	24	84	92	61	87		4		4	5			0.8	3	0.03	0.11	1.5			8
Barley Hay	90	57	57	25	57	60.89	9		28	37	65	98	2.1	8	0.30	0.28	1.6		0.19	25
Barley Silage	35	59	58	26	58		12	22	34	37	58	61	3.0	9	0.46	0.30	2.4		0.22	28
Barley Silage Mature	35	58	58	26	58		12	25	30	34	50	61	3.5	9	0.30	0.20	1.5		0.15	25
Barley Straw	90	44	44	1	43	43.98	4	70	42	55	78	100	1.9	7	0.32	0.08	2.2	0.67	0.16	7
Barley Grain	89	84	92	61	87		12	28	5	7	20	34	2.1	3	0.06	0.38	0.6	0.18	0.16	23
Barley Grain, Steam Flaked	85	90	100	70	100		12	39	5	7	20	30	2.1	3	0.06	0.35	0.6	0.18	0.16	23

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Barley Grain Steam Rolled	86	84	92	61	87		12	38	5	7	20	27	2.1	3	0.06	0.41	0.6	0.18	0.17	30
Barley Grain 2-row	87	84	92	61	87		12		6	8	24	34	2.3	2	0.05	0.31	0.6	0.18	0.17	
Barley Grain 6-row	87	84	92	61	87		11		6	8	24	34	2.2	3	0.05	0.36	0.6	0.18	0.15	
Barley Grain Lt. Wt. (42-44 lb/bu)	88	78	83	54	80		13	30	9	12	30	34	2.3	4						
Barley Feed Pearl Byproduct	90	74	78	49	76		15	25	12	15			3.9	5	0.05	0.45	0.7		0.06	
Barley Bran	91	59	59	28	59		12	28	21	27	36	6	4.3	7						
Barley Grain Screenings	89	71	74	46	73		12		9	11			2.6	4	0.35	0.33	0.9		0.15	
Beans Navy Cull	90	84	92	61	87	84.52	24	25	5	8	20	0	1.4	5	0.15	0.60	1.4	0.06	0.26	45
Beet Pulp Wet	17	77	82	53	79	75.09	9	35	20	25	45	30	0.7	5	0.65	0.08	0.9	0.40	0.22	21
Beet Pulp Dried	91	76	81	52	78	79.81	9	44	21	26	46	33	0.7	5	0.65	0.08	0.9	0.40	0.22	21
Beet Pulp Wet with Molasses	24	77	82	53	79		11	25	16	21	39	33	0.6	6	0.60	0.10	1.8		0.42	11
Beet Pulp Dried with Molasses	92	77	82	53	79	82.52	11	34	17	23	40	33	0.6	6	0.60	0.10	1.8		0.42	11
Beet Root (Sugar)	23	80	86	56	83		4		5	7	16		0.4	3						
Beet Tops (Sugar)	19	58	58	26	58		14		11	14	25	41	1.3	24	1.10	0.22	5.2	0.20	0.45	20
Beet Top Silage	25	52	52	16	51		12		12				2.0	32	1.38	0.22	5.7		0.57	20
Bermudagrass Coastal Dehydrated	90	62	63	33	63		16	40	26	29	40	10	3.8	7	0.40	0.25	1.8	0.72	0.23	18
Bermudagrass Coastal Hay	89	56	56	23	56	53.05	10	20	30	36	73	98	2.1	6	0.47	0.21	1.5	0.70	0.22	16
Bermudagrass Hay	89	53	53	18	53	50.79	10	18	29	37	72	98	1.9	8	0.46	0.20	1.5	0.70	0.25	31
Bermudagrass Silage	26	50	50	12	49		10	15	28	35	71	48	1.9	8	0.46	0.20	1.5	0.72	0.25	31
Birdsfoot Trefoil Fresh	22	66	68	38	67		21	20	21	31	47	41	4.4	9	1.78	0.25	2.6		0.25	31
Birdsfoot Trefoil Hay	89	57	57	25	57		16	22	31	38	50	92	2.2	8	1.73	0.24	1.8		0.25	28
Biuret	99	0	0	0	0		248	0	0	0	0	0	0.0	0	0.00	0.00	0.0	0.00	0.00	0
Blood Meal, Swine/Poultry	91	66	68	38	67		92	82	1	2	10	0	1.4	3	0.32	0.28	0.2	0.30	0.70	22
Bluegrass KY Fresh Early Bloom	36	69	71	43	70	75.62	15	20	27	32	60	41	3.9	7	0.37	0.30	1.9	0.42	0.19	25
Bluegrass Straw	93	45	45	3	44		6		40	50	78	90	1.1	6	0.20	0.10				
Bluestem Fresh Mature	61	50	50	12	49	56.82	6		34				2.5	5	0.40	0.12	0.8		0.05	28
Bread Byproduct	68	90	100	68	94		14	24	1	2	3	0	3.0	3	0.10	0.18	0.2	0.76	0.15	40
Brewers Grains Wet	23	85	93	62	88	62.66	26	52	13	21	45	18	7.5	4	0.30	0.58	0.1	0.15	0.32	78
Brewers Grains Dried	92	84	92	61	87	60.43	25	54	14	24	49	18	7.5	4	0.30	0.58	0.1	0.15	0.32	78
Brewers Yeast Dried	94	79	85	55	81		48		3				1.0	7	0.10	1.56	1.8		0.41	41
Bromegrass Fresh Immature	30	64	65	36	65	78.57	15	22	28	33	54	40	4.1	10	0.45	0.34	2.3		0.21	20

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Bromegrass Hay	89	55	55	21	55	62.19[e]	10	33	35	41	66	98	2.3	9	0.40	0.23	1.9	0.40	0.19	19
Bromegrass Haylage	35	57	57	25	57		11	26	36	44	69	61	2.5	8	0.38	0.30	2.0		0.20	19
Buckwheat Grain	88	75	79	50	77	72.27	12		13	17			2.8	2	0.11	0.36	0.5	0.05	0.16	10
Buttermilk Dried	92	88	98	65	91		34	0	5	0	0	0	5.0	10	1.44	1.00	0.9		0.09	44
Cactus, Prickly Pear	23	61	62	31	62		5		16	20	28		2.1	18	4.00	0.10	1.5		0.20	
Calcium Carbonate	99	0	0	0	0		0		0	0	0	0	0.0	99	38.50	0.04	0.1		0.00	0
Canarygrass Hay	91	53	53	18	53		9	26	32	34	67	98	2.7	8	0.38	0.25	2.7		0.14	18
Canola Meal, Solv. Ext.	90	72	75	47	74		41	30	11	19	29	23	2.0	8	0.74	1.14	1.1	0.07	0.78	68
Carrot Pulp	14	62	63	33	63		6		19	23	40	0	7.8	9						
Carrot Root Fresh	12	83	90	60	86	92.29	10		9	11	20	0	1.4	10	0.55	0.32	2.5	0.50	0.17	
Carrot Tops	16	73	77	48	75		13		18	23	45	41	3.8	15	1.94	0.19	1.9			
Cattle Manure Dried	92	38	40	0	36	30.58	15		35	42	55	0	2.5	14	1.15	1.20	0.6		1.78	240
Cheatgrass Fresh Immature	21	68	70	41	69		16		23				2.7	10	0.60	0.28				
Citrus Pulp Dried	90	78	83	54	80		7	38	13	20	21	33	2.9	7	1.81	0.12	0.8	0.04	0.08	14
Clover Ladino Fresh	19	69	71	43	70	73.22	25	20	14	33	35	41	4.8	11	1.27	0.38	2.4		0.20	20
Clover Ladino Hay	90	61	62	31	61	63.40	21	25	22	32	36	92	2.0	9	1.35	0.32	2.4	0.30	0.20	17
Clover Red Fresh	24	64	65	36	65		18	21	24	33	44	41	4.0	9	1.70	0.30	2.0	0.60	0.17	23
Clover Red Hay	88	55	55	21	55	58.33	15	28	30	39	51	92	2.5	8	1.50	0.25	1.7	0.32	0.17	17
Clover Sweet Hay	91	53	53	18	53		16	30	30	38	50	92	2.4	9	1.27	0.25	1.8	0.37	0.46	
Coconut Meal, Mech. Ext.	92	76	81	52	78	79.66	21	56	13	21	56	23	6.8	7	0.40	0.30	1.0	0.33	0.04	
Coffee Grounds	88	20	36	0	16		13		41	68	77	10	15.0	2	0.10	0.08				
Corn Whole Plant Pelleted	91	63	64	34	64		9	45	21	24	40	6	2.4	6	0.50	0.24	0.9		0.14	
Corn Fodder	80	65	66	37	66		9	45	25	29	48	100	2.4	7	0.50	0.25	0.9	0.20	0.14	
Corn Stover Mature (Stalks)	80	54	54	20	54		5	30	35	43	70	100	1.3	7	0.45	0.15	1.2	0.30	0.14	22
Corn Silage, Milk Stage	26	65	66	37	66		8	18	26	32	54	60	2.8	6	0.40	0.27	1.6		0.11	20
Corn Silage, Mature Well Eared	34	72	75	47	74	72.88	8	28	21	27	46	70	3.1	5	0.28	0.23	1.1	0.20	0.13	22
Corn Silage, Sweet Corn	24	65	66	37	66		11		20	32	57	60	5.0	5	0.24	0.26	1.2	0.17	0.16	39
Corn Grain, Whole	88	88	98	65	91	88.85	9	58	2	3	9	60	4.3	2	0.02	0.30	0.4	0.05	0.14	18
Corn Grain, Rolled	88	88	98	65	91		9	54	2	3	9	34	4.3	2	0.02	0.30	0.4	0.05	0.14	18
Corn Grain, Steam Flaked	85	93	104	71	97	95.44	9	59	2	3	9	40	4.1	2	0.02	0.27	0.4	0.05	0.14	18
Corn Grain, High Moisture	74	93	104	71	97	91.64	10	42	2	3	9	0	4.0	2	0.02	0.30	0.4	0.06	0.14	20
Corn Grain, High Oil	88	91	102	69	95		8	54	2	3	8	60	6.9	2	0.01	0.30	0.3	0.05	0.13	18

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Corn Grain, Hi-Lysine	92	87	96	64	90		12	58	4	4	11	60	4.4	2	0.03	0.24	0.4	0.05	0.11	18
Corn and Cob Meal	87	82	89	59	85	83.15	9	52	9	11	26	56	3.7	2	0.06	0.27	0.5	0.05	0.13	16
Corn Cobs	90	48	48	9	47	53.18	3	70	36	39	88	56	0.6	2	0.12	0.04	0.8		0.27	5
Corn Screenings	86	91	102	69	95		10	52	3	4	9	20	4.3	2	0.04	0.27	0.4	0.05	0.12	16
Corn Bran	91	76	81	52	78		11		10	17	51	0	6.3	3	0.04	0.15	0.1	0.13	0.08	18
Corn Germ, Full-fat	97	135	198	160	198		12	55	6	11	36	20	44.9	2	0.02	0.28	0.1	0.02	0.17	60
Corn Gluten Feed	90	80	86	56	83	78.47	22	25	9	12	38	36	3.2	7	0.11	0.84	1.3	0.25	0.47	84
Corn Gluten Meal 41% CP	91	85	93	62	88		46	63	5	9	32	23	3.2	3	0.13	0.55	0.2	0.07	0.62	35
Corn Gluten Meal 60% CP	91	89	99	67	93	75.29	67	65	3	6	11	23	2.5	2	0.06	0.54	0.2	0.10	0.90	40
Corn Cannery Waste	29	68	70	41	69		8	15	28	36	59	0	3.0	5	0.10	0.29	1.0		0.13	25
Cottonseed, Whole	91	95	107	73	99		23	38	27	37	47	100	19.4	5	0.16	0.64	1.0	0.06	0.24	34
Cottonseed, Whole, Delinted	90	95	107	73	99		24	39	19	28	40	100	22.9	5	0.12	0.54	1.2		0.24	36
Cottonseed, Whole, Extruded	92	87	98	67	91		26	50	32	44	53	33	9.5	5	0.17	0.68	1.3		0.24	38
Cotton Gin Trash (Burrs)	91	42	43	0	40		9		35	50	70	100	2.0	14	1.40	0.18	1.9		0.14	25
Cottonseed Hulls	90	45	45	3	44	44.30	5	45	48	70	87	100	1.8	3	0.15	0.08	1.0	0.02	0.05	10
Cottonseed Meal, Solv. Ext. 41% CP	90	77	82	53	79	72.85	47	42	13	18	25	23	1.5	7	0.22	1.23	1.6	0.05	0.44	66
Cottonseed Meal, Mech. Ext. 41% CP	92	79	85	55	81	71.71	46	50	13	19	31	23	5.0	7	0.21	1.18	1.6	0.05	0.42	64
Crab Waste Meal	91	29	37	0	30		32	65	11	13			3.0	43	15.00	1.88	0.5	1.63	0.27	107
Crambe Meal, Solv. Ext.	91	81	88	58	84		31	45	25	35	47	23	1.4	8	1.27	0.86	1.1	0.70	1.26	44
Crambe Meal, Mech. Ext.	92	88	98	65	91		28	50	24	33	42	25	17.0	7	1.22	0.78	1.0	0.65	1.18	41
Cranberry Pulp Meal	88	49	49	11	48		7		26	47	54	33	15.7	2						
Crawfish Waste Meal	94	25	36	0	29		35	74	12	15				42	13.10	0.85				
Curacao Phosphate	99	0	0	0	0		0		0	0	0	0	0.0	95	34.00	15.00				
Defluorinated Phosphate	99	0	0	0	0		0		0	0	0	0	0.0	95	32.60	18.07	1.0			100
Diammonium Phosphate	98	0	0	0	0		115	0	0	0	0	0	0.0	35	0.52	20.41	0.0		2.16	
Dicalcium Phosphate	96	0	0	0	0		0		0	0	0	0	0.0	94	22.00	18.65	0.1		1.00	70
Distillers Grains, Wet	25	91	102	69	95		28	52	8	18	40	4	9.6	5	0.10	0.70	1.0	0.20	0.60	95
Distillers Grain, Barley	90	75	79	50	77		30	56	16	20	44	4	8.5	4	0.15	0.67	1.0	0.18	0.43	50
Distillers Grain, Corn, Dry	91	95	106	72	99	76.86	30	58	8	16	44	4	9.5	4	0.09	0.75	0.9	0.14	0.70	65

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Distillers Grain, Corn, Wet	36	97	109	74	102		30	47	8	16	44	4	9.5	4	0.09	0.75	0.9	0.14	0.70	65
Distillers Grain, Corn with Solubles	89	98	111	76	103	81.50	30	54	8	16	38	4	11.9	6	0.20	0.75	0.9	0.18	0.80	85
Distillers Dried Solubles	93	87	96	64	91	79.45	31	47	4	7	22	4	13.0	8	0.35	1.20	1.8	0.28	1.10	91
Distillers Corn Stillage	7	92	103	70	96		22	55	8	10	21	0	8.1	5	0.14	0.72	0.2		0.60	60
Distillers Grain, Sorghum, Dry	91	84	92	61	87	72.85	33	62	13	20	44	4	10.0	4	0.20	0.68	0.3		0.50	50
Distillers Grain, Sorghum, Wet	35	86	95	63	89		33	55	13	19	43	4	10.0	4	0.20	0.68	0.3		0.50	50
Distillers Grain, Sorghum with Solubles	92	85	93	62	88		33	53	12	18	42	4	10.0	4	0.23	0.70	0.5		0.70	55
Elephant (Napier) Grass Hay, Chopped	92	55	55	21	54		9		24	46	63	85	2.0	10	0.35	0.30	1.3		0.10	
Fat, Animal, Poultry, Vegetable	99	195	285	230	285	80.08[f]	0		0	0	0	0	99.0	0	0.00	0.00	0.0			
Feather Meal Hydrolyzed	93	67	69	40	68		87	68	1	14	42	23	7.0	3	0.48	0.45	0.1	0.20	1.82	90
Fescue KY 31 Fresh	29	64	65	36	65		15	20	25	32	64	40	5.5	9	0.48	0.37	2.5		0.18	22
Fescue KY 31 Hay Early Bloom	88	60	60	30	60	53.57	18	22	25	31	64	98	6.6	8	0.48	0.36	2.6		0.27	24
Fescue KY 31 Hay Mature	88	52	52	16	51		11	30	30	42	73	98	5.0	6	0.45	0.26	1.7		0.14	22
Fescue (Red) Straw	94	43	44	0	41		4		41				1.1	6	0.00	0.06				
Fish Meal	90	74	78	49	76		66	60	1	2	12	10	9.0	20	5.55	3.15	0.7	0.76	0.80	130
Flax Seed Hulls	91	38	40	0	36		9		32	39	50	98	1.5	10						
Garbage Municipal Cooked	23	80	86	56	83		16		9	50	59	30	20.0	10	1.20	0.43	0.6	0.67		
Glycerol (Glycerin)	88	90	100	68	94		0	0	0	0	0	0	0.0	6				4.00		
Grain Screenings	90	65	66	37	66		14		14				5.5	9	0.25	0.34				30
Grain Dust	92	73	77	48	75		10		11				2.2	10	0.30	0.18				42
Grape Pomace Stemless	91	40	42	0	38	27.50	12	45	32	46	54	34	7.6	9	0.55	0.07	0.6	0.01		24
Grass Hay	88	58	58	26	58		10	30	33	41	63	98	3.0	6	0.60	0.21	2.0		0.20	28
Grass Silage	30	61	62	31	61		11	24	32	39	60	61	3.4	8	0.70	0.24	2.1		0.22	29
Guar Meal	90	72	75	47	74		39	34	16				3.9	5						
Hominy Feed	90	89	99	67	93		11	48	5	8	21	9	6.5	3	0.04	0.55	0.6	0.06	0.10	32
Hop Leaves	37	49	49	11	48		15		15				3.6	35	2.80	0.64				
Hop Vine Silage	30	53	53	18	53		15		21	24			3.1	20	3.30	0.37	1.8		0.22	44
Hops Spent	89	35	39	0	33		23		26	30			4.6	7	1.60	0.60				
Kelp Dried	91	32	38	0	29	54.67	7		7	10			0.5	39	2.72	0.31				
Kenaf Hay	92	48	48	9	47		10		31	44	56	98	2.9	12						
Kochia Fresh	29	55	55	21	55	65.11	16		23				1.2	18	1.10	0.30				
Kochia Hay	90	53	53	18	53		14		27				1.7	14	1.00	0.20				
Kudzu Hay	90	54	54	20	54		16		33				2.6	7	3.00	0.23				

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Lespedeza Fresh Early Bloom	25	60	60	30	60		16	50	32				2.0	10	1.20	0.24	1.1		0.21	
Lespedeza Hay	92	54	54	20	54		14	60	30				3.0	7	1.10	0.22	1.0		0.19	29
Limestone Ground	98	0	0	0	0		0	0	0	0	0	0	0.0	98	34.00	0.02			0.03	
Limestone Dolomitic Ground	99	0	0	0	0		0	0	0	0	0	0	0.0	98	22.30	0.04	0.4			
Linseed Meal, Solv. Ext.	91	77	82	53	79		38	36	10	18	25	23	1.7	6	0.43	0.91	1.5	0.04	0.47	60
Linseed Meal, Mech. Ext.	91	82	89	59	85		37	40	10	17	24	23	6.0	6	0.42	0.90	1.4	0.04	0.46	59
Meadow Hay	90	50	50	12	49	63.37	7	23	33	44	70	98	2.5	9	0.61	0.18	1.6		0.17	24
Meat Meal, Swine/Poultry	93	71	74	46	73		56	64	2	7	48	0	10.5	24	9.00	4.42	0.5	1.27	0.48	190
Meat and Bone Meal, Swine/Poultry	93	72	75	47	74		56	24	1	5	34	0	10.0	29	13.50	6.50				
Milk, Dry, Skim	94	87	96	64	90		36	0	0	0	0	0	0.9	8	1.36	1.09	1.7	0.96	0.34	41
Mint Slug Silage	27	55	55	21	55		14		24				1.8	16	1.10	0.57				
Molasses Beet	77	75	79	50	77	91.95	8	0	0	0	0	0	0.2	12	0.14	0.03	6.0	1.64	0.60	18
Molasses Cane	77	74	78	49	76	86.63	6	0	0	0	0	0	0.5	14	0.95	0.09	4.2	2.30	0.68	15
Molasses Cane Dried	94	74	78	49	76	82.12	9	0	2	3	7	0	0.3	14	1.10	0.15	3.6	3.00		30
Molasses, Cond. Fermentation Solubles	43	69	71	43	70		16	0	0	0	0	0	1.0	26	2.12	0.14	7.5	2.73	0.93	30
Molasses Citrus	65	75	79	50	77	84.11	9	0	0	0	0	0	0.3	8	1.84	0.15	0.2	0.11	0.23	137
Molasses Wood, Hemicellulose	61	70	73	44	71		1	0	1	2	4	0	0.6	7	1.10	0.10	0.1		0.05	
Monoammonium Phosphate	98	0	0	0	0		70	0	0	0	0	0	0.0	24	0.30	24.70	0.0		1.42	81
Mono-Dicalcium Phosphate	97	0	0	0	0		0		0	0	0	0	0.0	94	16.70	21.10	0.1		1.20	70
Oat Hay	90	54	54	20	54	59.36	10	25	31	39	63	98	2.3	8	0.40	0.27	1.6	0.42	0.21	28
Oat Silage	35	60	60	30	60	64.00[g]	12	21	31	39	59	61	3.2	10	0.34	0.30	2.4	0.50	0.25	27
Oat Straw	91	48	48	9	47	49.64	4	40	41	48	73	98	2.3	8	0.24	0.07	2.5	0.78	0.22	6
Oat Grain	89	76	81	52	78	75.63	13	18	11	15	28	34	5.0	4	0.05	0.41	0.5	0.11	0.20	40
Oat Grain, Steam Flaked	84	88	98	65	91		13	26	11	15	30	32	4.9	4	0.05	0.37	0.5	0.11	0.20	40
Oat Groats	91	91	102	69	95	88.29	18	15	3				6.6	2	0.08	0.47	0.4	0.10	0.20	
Oat Middlings	90	91	102	69	95		16	20	4	6			6.0	3	0.07	0.48	0.5		0.23	
Oat Mill Byproduct	89	33	38	0	30		7		27	37			2.4	6	0.13	0.22	0.6		0.24	
Oat Hulls	93	38	40	0	36	38.39	4	25	33	41	75	90	1.6	7	0.16	0.15	0.6	0.08	0.14	31
Orange Pulp Dried	89	79	85	55	81		9		9	16	20	33	1.8	4	0.71	0.11	0.6		0.05	
Orchardgrass Fresh Early Bloom	24	65	66	37	66	60.13	14	23	30	32	54	41	4.0	9	0.33	0.39	2.7	0.08	0.20	21
Orchardgrass Hay	88	59	59	28	59	64.29[h]	10	27	34	40	67	98	3.3	8	0.32	0.30	2.6	0.41	0.20	26
Pea Vine Hay	89	59	59	28	59		11		32	50	62	92	2.0	7	1.25	0.24	1.3		0.20	20

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
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Pea Vine Silage	25	58	58	26	58		16		29	44	55	61	3.3	8	1.25	0.28	1.6		0.29	32
Pea Vine Straw	89	51	51	14	50	49.62	7		41	49	72	98	1.4	7	0.75	0.13	1.1		0.15	
Peas Cull	88	85	93	62	88		23	22	7	9	12	0	1.4	4	0.14	0.46	1.1	0.06	0.26	30
Peanut Hulls	91	22	36	0	18	23.17	7		63	65	74	98	1.5	5	0.20	0.07	0.9			
Peanut Meal, Solv. Ext.	91	77	82	53	79	71.90	51	27	9	16	27	23	2.5	6	0.26	0.62	1.1	0.03	0.30	38
Peanut Skins	92	0	0	0	0		17		13	20	28	0	22.0	3	0.19	0.20				
Pearl Millet Grain	87	82	89	59	85	68.04	13		2	6	18	34	4.5	3	0.03	0.36	0.5			
Pineapple Greenchop	17	47	47	7	46		8		24	35	64	41	2.4	7	0.28	0.08				
Pineapple Bran	89	71	74	46	73	72.43	5		20	33	66	20	1.5	3	0.26	0.12				
Pineapple Presscake	21	71	74	46	73		5		24	35	69	20	0.8	3	0.25	0.09				
Potato Vine Silage	15	59	59	28	59		15		26				3.7	19	2.10	0.29	4.0		0.37	
Potatoes Cull	21	80	86	56	83		10	0	2	3	4	0	0.4	5	0.03	0.24	2.2	0.30	0.09	
Potato Waste Wet	14	82	89	59	85		7	0	9	11	18	0	1.5	3	0.16	0.25	1.2	0.36	0.11	12
Potato Waste Dried	89	85	93	62	88	95.85	8	0	7	9	15	0	0.5	5	0.16	0.25	1.2	0.39	0.11	12
Potato Waste Wet with Lime	17	80	86	56	83		5	0	10	12	16	0	0.3	9	4.20	0.18				
Potato Waste Filter Cake	14	77	82	53	79		5	0	2				7.7	3	0.10	0.19	0.2			
Poultry Byproduct Meal	93	79	85	55	81		62	49	2				14.5	17	4.00	2.25	0.5	0.58	0.56	129
Poultry Manure Dried	89	38	40	0	36	67.83	28	22	13	15	35	0	2.1	33	10.20	2.80	2.3	1.05	0.20	520
Prairie Hay	91	50	50	12	49	55.53	7	37	34	47	67	98	2.0	8	0.40	0.15	1.1	0.06	0.06	34
Pumpkins, Cull	11	80	86	56	83		15		14	21	30	0	8.9	9	0.24	0.43	3.3			
Rice Straw	91	40	42	0	38	51.16	4		38	47	72	100	1.4	13	0.23	0.08	1.2		0.11	
Rice Straw Ammoniated	87	45	45	3	44		9		39	53	68	100	1.3	12	0.25	0.08	1.1		0.11	
Rice Grain	89	79	85	55	81	83.86	8	30	10	12	16	34	1.9	5	0.07	0.32	0.4	0.09	0.05	17
Rice Polishings	90	90	100	68	94		14		4	5			14.0	9	0.05	1.34	1.2	0.12	0.19	28
Rice Bran	91	71	74	46	73	66.64	14	30	13	18	24	0	16.0	11	0.07	1.70	1.8	0.09	0.19	40
Rice Hulls	92	13	35	0	8	15.91	3	45	44	70	81	90	0.9	20	0.12	0.07	0.5	0.08	0.08	24
Rice Mill Byproduct	91	39	41	0	37		7		32	50	60	0	5.7	19	0.25	0.48	2.2		0.30	31
Rye Grass Hay	90	58	58	26	58	66.07[i]	10	30	33	38	65	98	3.3	8	0.45	0.30	2.2		0.18	27
Rye Grass Silage	32	59	59	28	59		14	25	22	37	59	61	3.3	8	0.43	0.38	2.9	0.73	0.23	29
Rye Straw	89	44	44	1	43	33.72	4		44	55	71	100	1.5	6	0.24	0.09	1.0	0.24	0.11	
Rye Grain	89	80	86	56	83	84.83	14	20	3	9	19	34	2.5	3	0.07	0.55	0.5	0.03	0.17	33
Safflower Meal, Solv. Ext.	91	56	56	23	56	57.72	24		33	41	57	36	1.3	6	0.35	0.79	0.9	0.21	0.23	65
Safflower Meal Dehulled, Solv. Ext.	91	75	79	50	77	70.55	47		11	20	27	30	0.8	7	0.38	1.50	1.2	0.18	0.22	36
Safflower Hulls	91	14	35	0	34		4		58	73	90	100	3.7	2						
Sagebrush Fresh	50	50	50	12	49	59.04[j]	13		25	30	38		9.2	10	1.00	0.25			0.22	
Sanfoin Hay	88	61	62	31	62		14	60	24				3.1	9						

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Shrimp Waste Meal	90	48	48	9	47		50	60	11				5.5	25	8.50	1.75		1.15		
Sodium Tripolyphosphate	96	0	0	0	0		0		0	0	0	0	0.0	96	0.00	25.98	0.0		0.00	
Sorghum Stover	87	54	54	20	54		5		33	41	65	100	1.8	10	0.50	0.12	1.2			
Sorghum Silage	32	59	59	28	59	65.58	9	25	27	38	59	70	2.7	6	0.48	0.21	1.7	0.45	0.11	30
Sorghum Grain (Milo), Ground	89	82	89	59	85		11	55	3	6	15	5	3.1	2	0.04	0.32	0.4	0.10	0.14	18
Sorghum Grain (Milo), Flaked	82	90	100	68	94		11	62	3	6	15	38	3.1	2	0.04	0.28	0.4	0.10	0.14	18
Soybean Hay	89	52	52	16	51	54.10	16		33	40	55	92	3.5	8	1.28	0.29	1.0	0.15	0.24	24
Soybean Straw	88	42	43	0	40	45.98	5		44	54	70	100	1.4	6	1.59	0.06	0.6		0.26	
Soybeans Whole	88	92	103	70	96		41	28	8	11	15	100	18.8	5	0.27	0.64	1.9	0.03	0.34	56
Soybeans Whole, Extruded	88	93	104	71	97		40	35	9	11	15	100	18.8	5	0.27	0.64	2.0	0.03	0.34	56
Soybeans Whole, Roasted	88	93	104	71	97		40	48	9	11	15	100	18.8	5	0.27	0.64	2.0	0.03	0.34	56
Soybean Hulls	90	77	82	52	79	66.86	13	28	39	48	62	28	2.3	5	0.60	0.19	1.3	0.02	0.12	38
Soybean Meal, Solv. Ext. 44% CP	89	84	92	61	87	79.50	49	35	7	10	15	23	1.5	7	0.36	0.70	2.2	0.07	0.41	62
Soybean Meal, Solv. Ext. 49% CP	89	87	96	64	90		54	36	4	6	8	23	1.3	7	0.28	0.71	2.2	0.08	0.45	61
Soybean Mill Feed	90	50	50	12	49		15		36	46			1.9	6	0.46	0.19	1.7		0.07	
Spelt Grain	88	75	79	50	77	77.18	13	27	10	17	21	34	2.1	4	0.04	0.40	0.4		0.15	47
Sudangrass Fresh Immature	18	70	73	44	71	73.27	17		23	29	55	41	3.9	9	0.46	0.36	2.0		0.11	24
Sudangrass Hay	88	57	57	25	57	62.67	9	30	36	43	67	98	1.8	10	0.50	0.22	2.2	0.80	0.12	26
Sudangrass Silage	31	58	58	26	58	60.29	10	28	30	42	64	61	3.1	10	0.58	0.27	2.4	0.52	0.14	29
Sunflower Meal, Solv. Ext.	92	65	66	37	66	44.89	40	27	18	22	36	23	2.8	8	0.44	0.97	1.1	0.15	0.33	55
Sunflower Meal with Hulls	91	57	57	25	57		31	35	27	32	44	37	2.4	7	0.40	1.03	1.0		0.30	85
Sunflower Seed Hulls	90	40	42	0	38		4	65	52	63	73	90	2.2	3	0.00	0.11	0.2		0.19	200
Sugar Cane Bagasse	91	39	41	0	37	52.15	1		49	60	86	100	0.6	4	0.90	0.29	0.5		0.10	
Tapioca Meal, Cassava Byproduct	89	82	89	59	85		1		5	8	34		0.8	3	0.03	0.05				
Timothy Fresh Pre-Bloom	26	64	65	36	65		11	20	31	36	59	41	3.8	7	0.40	0.28	1.9	0.57	0.15	28
Timothy Hay Early Bloom	88	59	59	28	59	60.75	11	22	32	39	63	98	2.7	6	0.58	0.26	1.9	0.51	0.21	30
Timothy Hay Full Bloom	88	57	57	25	57	58.68	8	30	34	40	65	98	2.6	5	0.43	0.20	1.8	0.62	0.13	25
Timothy Silage	34	59	59	28	59	59.32	10	25	34	45	70	61	3.4	7	0.50	0.27	1.7		0.15	
Tomatoes	6	69	71	43	70		16		9	11			4.0	6	0.14	0.35	4.2			
Tomato Pomace Dried	92	64	65	36	65	53.98	23		26	50	55	34	10.6	6	0.43	0.59	3.6			
Triticale Hay	90	56	56	23	56		10		34	41	69	98			0.30	0.26	2.3			25

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Triticale Silage	34	58	58	26	58		14		30	39	56	61	3.6		0.58	0.34	2.7		0.28	36
Triticale Grain	89	85	93	62	88	83.82	14	25	4	5	22	34	2.4	2	0.07	0.39	0.5		0.17	37
Turnip Tops (Purple)	18	68	70	41	69		18		10	13			2.6	14	3.10	0.40	3.0	1.80	0.27	
Turnip Roots	9	86	95	63	89	92.94	12	0	11	34	44	40	1.6	9	0.65	0.31	3.1	0.65	0.43	40
Urea 46% N	99	0	0	0	0		288	0	0	0	0	0	0.0	0	0.00	0.00	0.0	0.00	0.00	0
Vetch Hay	89	58	58	26	58	59.44	18	14	30	33	48	92	1.8	8	1.25	0.34	2.4		0.13	
Wheat Fresh, Pasture	21	71	74	46	73	76.07	20	16	18	30	50	41	4.0	13	0.35	0.36	3.1	0.67	0.22	
Wheat Hay	90	57	57	25	57	62.73	9	25	29	38	66	98	2.0	8	0.21	0.22	1.4	0.50	0.19	23
Wheat Silage	33	59	59	28	59	63.99	12	21	28	37	62	61	3.2	8	0.40	0.28	2.1	0.50	0.21	27
Wheat Straw	91	43	44	0	41	45.77	3	60	43	57	81	98	1.8	8	0.17	0.06	1.3	0.32	0.17	6
Wheat Straw Ammoniated	85	50	50	12	49		9	25	40	55	76	98	1.5	9	0.15	0.05	1.3	0.30	0.16	6
Wheat Grain	89	88	98	65	91	86.45[k]	14	23	3	4	12	0	2.3	2	0.05	0.43	0.4	0.09	0.15	40
Wheat Grain Hard	89	88	98	65	91	88.54[l]	14	28	3	6	14	0	2.0	2	0.05	0.43	0.5		0.16	45
Wheat Grain Soft	89	88	98	65	91	89.96[m]	12	23	3	4	12	0	2.0	2	0.06	0.40	0.4		0.15	30
Wheat Grain, Steam Flaked	85	91	102	69	95		14	29	3	4	12	0	2.3	2	0.05	0.39	0.4		0.15	40
Wheat Grain Sprouted	86	88	98	65	91		12	18	3	4	13	0	2.0	2	0.04	0.36	0.4		0.17	45
Wheat Bran	89	70	73	44	71	71.16	17	28	11	14	46	4	4.4	7	0.13	1.32	1.4	0.05	0.24	96
Wheat Middlings	89	80	86	56	83		18	22	8	11	36	2	4.7	5	0.14	1.00	1.3	0.05	0.20	98
Wheat Mill Run	90	76	81	52	78	79.11	17	28	9	12	37	0	4.5	6	0.11	1.10	1.2	0.07	0.22	90
Wheat Shorts	89	78	83	54	80		19	25	8	10	30	0	5.3	5	0.10	0.93	1.1	0.08	0.20	118
Wheatgrass Crested Fresh Early Bloom	37	60	60	30	60	79.78	11	25	26	28	50	41	1.6	7	0.46	0.32	2.4			
Wheatgrass Crested Fresh Full Bloom	50	55	55	21	55	65.89	10	33	33	36	65	41	1.6	7	0.39	0.28	2.1			
Wheatgrass Crested Hay	92	54	54	20	54	56.51	10	33	33	36	65	98	2.4	7	0.33	0.20	2.0			32
Whey Dried	94	82	89	59	85	91.47[n]	14	15	0	0	0	0	0.9	10	0.98	0.88	1.3	1.20	0.92	10
Yeast, Brewer's	92	79	85	55	81	73.76	47	30	3	4		0	0.9	7	0.13	1.49	1.8			

DM = Dry matter
TDN = Total digestible nutrients
NEm = Net energy for maintenance
NEg = Net energy for growth
NEl = Net energy for lactation
Mcal = Megacalories
cwt = Centum weight (hundredweight)
DE =Digestible energy
GE = Gross energy
CP =Crude protein
UIP = Undegradable intake protein
CF = Crude fiber
ADF = Acid detergent fiber
NDF = Neutral detergent fiber
eNDF = effective neutral detergent fiber
EE = Ether extract
ASH = Ash
Ca = Calcium
P = Phosphorous
K = Potassium
Cl = Chlorine
S = Sulfur
Zn = Zinc
ppm = parts per million

[a] DE (% of GE) values from Ewan (1989)
[b] Average of fresh, late vegetative; fresh, early bloom; fresh, midbloom; fresh, full bloom
[c] Average of silage wilted – early bloom; silage wilted – midbloom; silage wilted – full bloom
[d] Average of silage wilted – early bloom; silage wilted – midbloom; silage wilted – full bloom

[e] Average of hay – sun-cured, late vegetative; hay – sun-cured, late bloom
[f] Average of fat, animal poultry; oil, vegetable
[g] Average of silage, late vegetative; silage, dough stage
[h] Average of hay, sun-cured, early bloom; hay, sun-cured, late bloom
[i] Average of ryegrass, Italian *Lolium multiflorum*: hay, sun-cured, late vegetative; hay, sun-cured, early bloom; average of ryegrass, perennial *Lolium perenne*: hay, sun-cured
[j] Average of sagebrush, big *Artemisia* tridentate: browse, fresh, stem-cured; sagebrush, bud *Artemisia spinescens*: browse, fresh, early vegetative; browse, fresh, late vegetative; and sagebrush, fringed *Artemisia frigida*: browse, fresh, midbloom; browse, fresh, mature
[k] Average of wheat, *Durum Triticum* durum and wheat *Triticum aestivum* grain
[l] Average of grain, hard red spring; grain, hard winter
[m] Average of grain, soft red winter; grain, soft white winter; grain, soft white winter, pacific coast
[n] Average of dehydrated (cattle) and low lactose, low lactose, dehydrated (dried whey product)(cattle)

Appendix 5-C: Estimation Methods for Ammonia Emissions from Manure Management Systems

This appendix presents methods for estimating NH_3 from manure management systems. NH_3, although not a GHG, is emitted in large quantities from animal housing and manure management systems and is an indirect precursor to nitrous oxide (N_2O) emissions as well as an environmental concern.

5-C.1 Method for Estimating Ammonia Emissions Using Equations from Integrated Farm System Model

Ammonia

- Method is a function of the surface area of the storage unit, resistance to mass transfer, ambient air velocity, total NH_3 and organic nitrogen content, rate of organic nitrogen transformation to total ammoniacal nitrogen, and manure temperature as defined by Rotz et al. (2011b).
- Ammonia and organic nitrogen content can be obtained from sampling and lab testing.

Ammonia emissions from manure storage are mainly from total ammoniacal nitrogen (TAN). For many animal confinement systems, it has been reported that most of the urea in manure has been converted to TAN and lost as NH_3 by the time manure is transferred to storage (Rotz et al., 2011b); therefore, only organic nitrogen in the manure at the storage stage, which is mineralized to TAN, is used to estimate NH_3 release. There are four main steps related to NH_3 release to the atmosphere: diffusion, dissociation, aqueous to gas partitioning, and mass transport away from the manure surface (Rotz et al., 2011b). For solid manure, diffusion through the manure is a main constraint to the emission rate. For liquid manure, NH_3 emissions are a function of the overall mass transfer rate and the difference in the NH_3 concentration between the lagoon and the surrounding atmosphere.

5-C.1.1 Rationale for Selected Method

Ammonia emissions from temporary stack and long term stockpiles, aerobic lagoons, anaerobic lagoons, runoff holding ponds, and storage tanks can be calculated using equations from the DairyGEM Model (a subset of the Integrated Farm System Model) (Rotz et al., 2011b). The equations from Rotz et al. are the only available methods for estimating NH_3 emissions from these systems and best describes the quantitative relationship amongst activity data at the entity level.

5-C.1.2 Activity Data

In order to estimate the daily NH_3 emission from temporary stack, long-term stockpiles, anaerobic lagoons, runoff holding ponds, and storage tanks, the following information is needed:

- Total nitrogen content of manure
- Manure total NH_3-N content
- Surface area of manure pile
- Temperatures (local ambient temperature and manure temperature)
- Local ambient air velocity
- For aerobic lagoons, the pH of the lagoon is also needed.

The timing of measurements can be based on dietary changes or seasonal timeframe, which is decided by individual farm entity. However, due to the dynamic nature of manure piles causing the changes of the variables, frequent measurements of manure characteristics are recommended to ensure accuracy of the estimation.

5-C.1.3 Ancillary Data

The ancillary data used to estimate NH_3 emission for temporary storage are kinematic viscosity of air, mass diffusivity of NH_3, and resistance to mass transfer. The kinematic viscosity of air at standard atmospheric pressure is listed in Table 5-C-1. The mass diffusivity of NH_3 is obtained from references (Paul and Watson, 1966; Baker, 1969) and listed in Table 5-C-2. The resistance to mass transfer for different solid manure storages are obtained from the DairyGEM model (Rotz et al., 2011a).

5-C.2 Method for Ammonia Emissions from Temporary Stack, Long-Term Stockpile, Anaerobic Lagoons/Runoff Holding Ponds/Storage Tanks, and Aerobic Lagoons

Temporary Stack, Long-Term Stockpile, and Anaerobic Lagoons/Runoff Holding Ponds/Storage Tanks

As indicated in Equation 5-C-1, NH_3 emissions are a function of the overall mass transfer rate and the difference in NH_3 concentration between the manure and surrounding atmosphere. The mean ambient air NH_3 concentration is 1.3 μg/m^3 based on passive measurements from 35 locations across 24 States in the U.S. with one year or more of measurements (Ammonia Monitoring Network, National Atmospheric Deposition Program). The Henry's Law constant is used to define the ratio of NH_3 concentration in a solution in equilibrium with gaseous NH_3 concentration in air and is exponentially related to temperature.

Equation 5-C-1: Ammonia Emissions from Temporary Stack, Long Term Stockpiles, and Anaerobic Lagoons/Runoff Holding Ponds/Storage Tanks

$$\mathbf{E_{NH_3} = 24 \times 3600 \times A_{surface} \times K \times (TAN_m - H \times TAN_a)}$$

Where:

E_{NH3} = NH_3 emissions per day (kg NH_3 day^{-1})

24 = Hours per day (hr day^{-1})

3,600 = Seconds per hour (s hr^{-1})

$A_{surface}$ = Footprint of manure storage (m^2) × shape factor[b]

K = Overall mass transfer coefficient (m s^{-1}) as defined in Equation 5-C-3

TAN_m = Total ammoniacal nitrogen in the manure (kg m^{-3})

TAN_a = NH_3 concentration in ambient air[a] (kg m^{-3})

H = Henry's Law constant as defined in Equation 5-C-2

[a] Ammonia concentration in ambient air can be obtained from National Atmospheric Deposition Program (nadp.sws.uiuc.edu/amon/).
[b] Shape factors (ℝ) are listed in Appendix 5-D.

Equation 5-C-2 describes the calculation for Henry's Law Constant. The manure temperature is calculated as the average ambient temperature over the previous 10 days.

Equation 5-C-2: Calculation Henry's Law Constant

$$\mathbf{H} = \frac{\mathbf{T}}{\mathbf{0.2138}} \times \mathbf{10}^{\left(\frac{\mathbf{1825}}{\mathbf{T}} - \mathbf{6.123}\right)}$$

Where:

H = Henry's Law constant for NH_3 (aqueous to gas)

T = Manure temperature (Kelvin degree)

The overall mass transfer coefficient is expressed as the reciprocal of the overall effective resistance of the manure. The mass transfer coefficient through gaseous phase on the top of manure is calculated using Equation 5-C-3. The resistance to mass transfer is calculated in Equation 5-C-5. It has been reported that the mass transfer coefficient through manure has relatively little effect on the mass transfer of NH_3 (Ni, 1999) and thus the $1/K_l$ is considered negligible in the following equation.

Equation 5-C-3: Overall Mass Transfer Coefficient

$$\mathbf{K} = \frac{\mathbf{1}}{\left(\frac{\mathbf{H}}{\mathbf{K_g}} + \frac{\mathbf{1}}{\mathbf{K_l}} + \mathbf{R_m}\right)}$$

Where:

K = Overall mass transfer coefficient (m s^{-1})

H = Henry's Law constant for NH_3 (aqueous to gas)

R_m = Resistance to mass transfer (s m^{-1})

K_g = Mass transfer coefficient through gaseous phase on the top of manure (m s^{-1})

K_l = Mass transfer coefficient through manure (m s^{-1})

The mass transfer coefficient through gaseous phase (Equation 5-C-4) is estimated from the air friction velocity and Schmidt number of air. The Turbulent Schmidt number is dependent on the characteristics of the gas and the scales of atmospheric turbulence. Since turbulence is highly dependent on many complex interactions, the Turbulent Schmidt number was approximated by only accounting for the gas characteristics. These characteristics are expressed in the molecular Schmidt number, defined as SC =ν/D, where ν is the kinematic viscosity of air (m^2 s^{-1}), and D is the mass diffusivity of NH_3 (m^2 s^{-1}). In order to calculate Schmidt number, the dynamic viscosity of air, the density of the air, and the mass diffusivity of NH_3 are given based on air temperature in Table 5-C-1 and Table 5-C-2.

Equation 5-C-4: Calculating Mass Transfer Coefficient through Gaseous Phase

$$\mathbf{K_g = 0.001 + 0.0462 \times (0.02 \times V_a^{1.5}) \times (SC)^{-0.67}}$$

Where:

K_g = Mass transfer coefficient through gaseous phase on the top of manure (m s^{-1})

V_a = Ambient air velocity (m s^{-1}) that can be obtained from National Weather Service by searching the target location

SC = Turbulent Schmidt number of NH_3 in the air above manure surface (dimensionless)

Table 5-C-1: Kinematic Viscosity of Air at Different Temperature at Standard Atmospheric Pressure

Temperature (°C)	Kinematic Viscosity (m^2/s) x 10^{-5}
-40	1.04
-20	1.17
0	1.32
5	1.36
10	1.41
15	1.47
20	1.51
25	1.56
30	1.60
40	1.66
50	1.76

Source: White (1999).

Table 5-C-2: Mass Diffusivity of Ammonia at Standard Atmospheric Pressure

Temperature (°C)	Diffusivity of Ammonia (m^2/s) x 10^{-4}
-40	0.106
0	0.110
30	0.200
40	0.209
50	0.233

Source: Paul and Watson (1966) and Baker (1969).

The mass transfer coefficient through manure has little effect on the mass transfer of NH_3, so it is negligible. The resistance to mass transfer is the sum of the resistance through the manure and the resistance of cover materials over the manure (Equation 5-C-5). The values for resistance to mass transfer through the manure and resistance to mass transfer through the cover are listed in Table 5-C-3 for temporary stack and long-term stockpile and in Table 5-C-4 for anaerobic lagoons, runoff holding ponds, and storage tanks.

Equation 5-C-5: Calculation of Resistance to Mass Transfer

$$\mathbf{R_m = R_S + R_C}$$

Where:

R_m = Resistance to mass transfer (s m^{-1})

R_s = Resistance to mass transfer through the manure (s m^{-1})

R_c = Resistance to mass transfer through the cover (s m^{-1})

Table 5-C-3: Resistance to Mass Transfer for Solid Manure Storage

Type of Manure Storage	R_s (s m^{-1})	R_c (s m^{-1})
Uncovered solid manure (dry matter >15%)	3×10^5	0
Covered solid manure (dry matter >15%)	3×10^5	2×10^5
Uncovered slurry manure (dry mater, 10-15%)	2×10^5	0

Type of Manure Storage	R_s (s m^{-1})	R_c (s m^{-1})
Covered slurry manure (dry mater, 10-15%)	2×10^5	2×10^5

Source: Rotz et al. (2011b).

Table 5-C-4: Resistance to Mass Transfer For Anaerobic Lagoons, Runoff Holding Ponds, and Storage Tanks

Type of Cover	R_s (s m^{-1})	R_c (s m^{-1})
Uncovered liquid manure	0	0
Covered liquid manure	0	2×10^5

Source: Rotz et al. (2011a).

Aerobic Lagoons

The method for estimating NH_3 emissions from aerobic lagoons (Equation 5-C-6) is similar to that for stockpiles and anaerobic lagoons but accounts for the concentration of NH_3 in the liquid.

Equation 5-C-6: Resistance to Mass Transfer for Solid Manure Storage (Rotz et al., 2011b)

$$\mathbf{E_{NH_3} = 24 \times 3600 \times K \times A_{surface} \times NH_3}$$

Where:

E_{NH3} = NH_3 emissions per day (kg day^{-1})

24 = Hours per day (hr day^{-1})

3,600 = Seconds per hour (s h^{-1})

K = Overall mass transfer coefficient (m s^{-1})

$A_{surface}$ = Surface area of lagoon (m^2)

NH_3 = Concentration in the liquid (kg m^{-3})

The overall mass transfer coefficient is calculated using Equation 5-C-3 with resistance to mass transfer assumed to be zero. Henry's Law Constant is calculated using Equation 5-C-2 and the mass transfer coefficient through a gaseous phase is calculated using **Equation 5-C-4**. The mass transfer through the liquid film layer is calculated using Equation 5-C-7.

Equation 5-C-7: Calculating the Mass Transfer Coefficient through the Liquid Film Layer

$$K_l = 1.417 \times 10^{-12} \times T^4$$

Where:

K_l = Mass transfer coefficient through the liquid film layer (m s^{-1})

T = Manure temperature (Kelvin)

Equation 5-C-8 describes the estimation method for NH_3 concentration in the liquid. The NH_3 fraction of TAN in the lagoon liquid is a function of pH and a dissociation constant according to Equation 5-C-9.

Equation 5-C-8: Calculating the Ammonia Concentration in the Liquid

$$\mathbf{NH_3 = F \times TAN}$$

Where:

NH_3 = Concentration in the liquid (kg m^{-3})

F = NH_3 of TAN in the lagoon liquid

TAN = Total ammonia nitrogen in the manure liquid (kg m^{-3})

Equation 5-C-9: Calculating the Ammonia Fraction of TAN in the Lagoon Liquid

$$\mathbf{F = \frac{1}{1 + \frac{10^{-pH}}{K_a}}}$$

Where:

F = NH_3 of TAN in the lagoon liquid

pH = Hydrogen ion concentration

K_a = Dissociation constant, where $K_a = 10^{\left(0.05 - \frac{2788}{T}\right)}$

T = Temperature (Kelvin)

5-C.3 Method for Estimating Ammonia Emissions from Composting Using IPCC Tier 2 Equations

Ammonia

- IPCC Tier 2 approach adjusted to estimate NH_3 emissions utilizing data on an NH_3 emission factor, total initial nitrogen, and dry manure.
- The NH_3 emission factor is obtained from a study of composting mixture of cattle and swine manure by Hellebrand and Kalk (2000).
- Nitrogen content can be obtained from sampling and lab testing.
- Method is the only readily available method.

Composting is the controlled aerobic decomposition of organic material into a stable, humus-like product (USDA NRCS, 2007). Eghball et al. (1997) reported that 19 to 45 percent of the nitrogen present in manure was lost during composting, with the majority of this presumably as NH_3.

5-C.3.1 Rationale for Selected Method

The IPCC method is adapted for estimating NH_3 emissions and incorporates NH_3 emission factors from a study of composting cattle and swine manure (Hellebrand and Kalk, 2000). The IPCC equation is the only available method for estimating NH_3 emissions from composting. This methodology best describes the quantitative relationship amongst activity data at the entity level.

5-C.3.2 Activity Data

In order to estimate the daily NH_3 emissions, the following information is needed:

- Total dry manure in the storage
- Total nitrogen in manure

The timing of measurements can be based on dietary changes or on a seasonal timeframe, which is decided by individual farm entity. However, due to the dynamic nature of manure storage causing changes in the variables, frequent measurements of manure characteristics (e.g., volatile solids, temperature, total dry manure) are recommended to improve accuracy of the estimation.

5-C.3.3 Ancillary Data

The ancillary data used to estimate NH_3 emission for manure composting is NH_3 emission factor (Hellebrand and Kalk, 2000).

5-C.4 Method for Ammonia Emissions from Composting

Ammonia emissions from composting are dependent on volatilization and mineralization after nitrification, decomposition of organic nitrogen compounds, or urea hydrolysis. An IPCC Tier 2 approach for estimating N_2O emissions is adapted to estimate NH_3 emissions from composting of solid manure. The NH_3 emission factor of 0.05 is obtained from a study of composting mixture of cattle and swine manure (Hellebrand and Kalk, 2000). Equation 5-C-10 provides the equations for estimating NH_3 emissions.

Equation 5-C-10: IPCC Tier 2 Approach for Calculating NH_3 Emissions from Composting of Solid Manure

$$\mathbf{E_{NH3} = m \times EF_{NH3} \times TN \times \frac{17}{14}}$$

Where:

E_{NH3} = NH_3 emissions per day (kg NH_3 day^{-1})

m = Total dry manure (kg day^{-1})

EF_{NH3} = NH_3 emission (loss) relative to total nitrogen in manure (kg NH_3-N (kg TN)$^{-1}$; =0.05)

TN = Total nitrogen in the initial (fresh) manure (kg TN (kg dry manure)$^{-1}$)

$\frac{17}{14}$ = Conversion of NH_3 to nitrogen

5-C.5 Uncertainty in Ammonia Emissions Estimates

Estimation methods from Rotz et al. (2011b) are used to estimate NH_3 emissions from temporary stack and long-term stockpiles and aerobic lagoons. Rotz et al. takes into account the amount of emissive surface area of the pile or lagoon. Given the difficulty of measuring the surface area of a manure pile, shape factors have been developed to approximate surface area based on general shape and footprint. These shape factors provide an estimate total surface area only; there is associated uncertainty based on the accurracy of the footprint measurements and how well the shape of the pile matches the shape factors defined.

The Rotz et al. equations require the NH_3 concentration in the ambient air on site. National data on ambient NH_3 concentrations are available from the National Atmospheric Deposition Program. The

Program provides ambient NH_3 concentrations from approximately 60 active monitoring sites across the country. Given the dearth of monitoring sites and the potentially long distances between the entity and the nearest measurement, there can be a large amount of uncertainty associated with the ambient air NH_3 concentrations used for estimating NH_3 emissions.

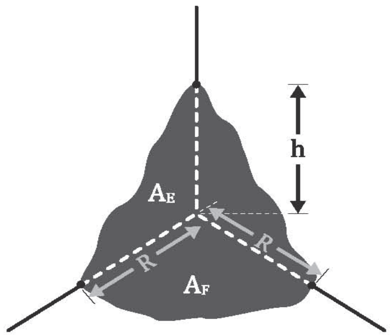
Table 5-C-5: Available Uncertainty Data for Ammonia Emissions Estimates

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
pH	pH	-	7.5			6.5	8.5	Expert Assessment
Total ammonia nitrogen in the manure – beef earthen lot	TAN	kg NH_3/m^3	0.1			0	0.02	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, leghorn pullets	TAN	kg NH_3/m^3	0.85			0.66	1.04	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, leghorn hen	TAN	kg NH_3/m^3	0.88			0.54	1.22	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, broiler	TAN	kg NH_3/m^3	0.75					ASABE (2005)
Ammonia concentration in the liquid – dairy lagoon effluent	NH_3	kg NH_3/m^3	0.08					ASABE (2005)
Ammonia concentration in the liquid – dairy slurry (liquid)	NH_3	kg NH_3/m^3	0.14					ASABE (2005)
Ammonia concentration in the liquid – Swine Finisher-Slurry wet-dry feeders	NH_3	kg NH_3/m^3	0.5					ASABE (2005)
Ammonia concentration in the liquid – Swine Slurry storage-dry feeders	NH_3	kg NH_3/m^3	0.34			0.19	0.49	ASABE (2005)
Ammonia concentration in the liquid – Swine flush building	NH_3	kg NH_3/m^3	0.14					ASABE (2005)
Ammonia concentration in the liquid – Swine agitated solids and water	NH_3	kg NH_3/m^3	0.05					ASABE (2005)
Ammonia concentration in the liquid – Swine Lagoon surface water	NH_3	kg NH_3/m^3	0.04					ASABE (2005)
Ammonia concentration in the liquid – Swine Lagoon sludge	NH_3	kg NH_3/m^3	0.07					ASABE (2005)
Composting – Ammonia emission (loss) relative to total nitrogen in manure	EF_{NH3}	kg NH_3-N/kg N	0.05					Hellebrand and Kalk (2000)

Appendix 5-D: Manure Management Systems Shape Factors ($\mathbb{R}$)

Factors can be applied to account for the differences in emissive surface areas for different shapes of manure piles. The equations provided below provide estimates for the surface area for common pile shapes; these estimates are applied for calculating NH_3 emissions from temporary stacks.

Figure 5-D-1: Equations for Calculating the Shape Factor for a 2-Sided Storage Bin with Quarter-Cone Pile



$$\mathbb{R} = \sqrt{\frac{h^2}{R^2} + 1}$$

Emitting Area, A_E

$$A_E = \frac{\pi R h}{4} \sqrt{1 + \left(\frac{R}{h}\right)^2}$$

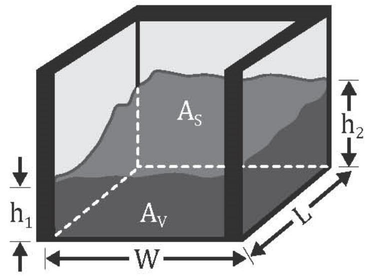
Footprint Area, A_F

$$A_F = \frac{\pi R^2}{4}$$

Limiting Cases

- If $h = R$, $\mathbb{R} = 1.4$
- If $R \gg h$, $\mathbb{R} = 1.0$

Figure 5-D-2: Equations for Calculating the Shape Factor for a 3-Sided Storage Bin



$$\mathbb{R} = \frac{A_E}{A_F} = \frac{h_1 + \sqrt{L^2 + (h_2 - h_1)^2}}{L}$$

Emitting Area, A_E

$$A_E = A_V + A_S$$

$$A_V = h_1 W$$

$$A_S = \sqrt{L^2 + (h_2 - h_1)^2}\ W$$

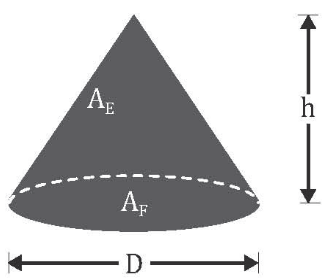
Footprint Area, A_F

$$A_F = LW$$

Limiting Cases

- If $h_1 = 0$, $\mathbb{R} = \dfrac{\sqrt{L^2 + h_2^2}}{L}$
- If $h_1 = h_2$, $\mathbb{R} = \dfrac{h_1 + L}{L} = 1 + \dfrac{h_1}{L}$
- If $0 < h_1 < h_2$, $\mathbb{R} = \dfrac{h_1 + \sqrt{L^2 + (h_2 - h_1)^2}}{L}$

Figure 5-D-3: Equations for Calculating the Shape Factor for a Conical Manure Pile



$$\mathbb{R} = \sqrt{\left(\frac{4h^2}{D^2} + 1\right)}$$

Emitting Area, A_E

$$A_E = \pi \frac{Dh}{2}\left[1 + \frac{D^2}{4h^2}\right]^{1/2}$$

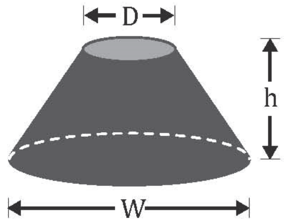
Footprint Area, A_F

$$A_F = \pi D^2/4$$

Limiting Cases

- If $D \gg h$, $\mathbb{R} = 1.0$
- If $D = 2h$, $\mathbb{R} = 1.4$

Figure 5-D-4: Equations for Calculating the Shape Factor for a Free-Standing, Truncated Conical Stack



$$\mathbb{R} = \frac{A_E}{A_F} = \frac{\left\{2\,h\,(W+D)\sqrt{1+\frac{(W-D)^2}{4h^2}}\right\} + D^2}{W^2}$$

Emitting Area, A_E

$$A_E = \left\{\frac{\pi\,h\,(W+D)}{2}\sqrt{1+\frac{(W-D)^2}{4h^2}}\right\} + \left\{\frac{\pi\,D^2}{4}\right\}$$

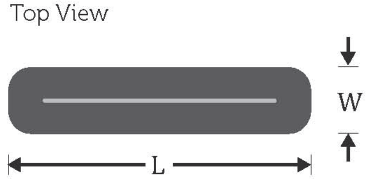
Footprint Area, A_F

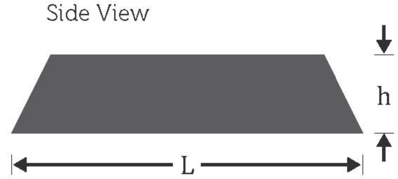
$$A_F = \pi\,\frac{W^2}{4}$$

Limiting Cases

- If $D = 0$, see "free-standing conical stack"
- If h is very small and $D \approx W$, $\mathbb{R} = 1.0$

Figure 5-D-5: Equations for Calculating the Shape Factor for a Windrow with Triangular Cross Section





$$\mathbb{R} = \sqrt{\left(\frac{4h^2}{W^2} + 1\right)}$$

Emitting Area, A_E

$$A_E = 2h\left[\frac{\pi W}{4} + L\right]\sqrt{\left(1 + \frac{W^2}{4h^2}\right)}$$

Footprint Area, A_F

$$A_F = W\left[\frac{\pi W}{4} + L\right]$$

Limiting Cases

- If $2h = W$, $\mathbb{R} = 1.4$
- If $W \gg h$, $\mathbb{R} = 1.0$

Note: Ratio $\mathbb{R} = \frac{A_E}{A_F}$

Does not depend on Windrow Length, L.

Appendix 5-E: Model Review: Review of Enteric Fermentation Models

A number of empirical and mechanistic models have been developed to estimate enteric CH_4 production (Table 5-E-1). Two of the factors that affect enteric CH_4 production to the greatest extent are diet composition and level of intake. Prediction equations and models constructed to predict enteric CH_4 are generally based on these factors. Most statistical equations developed to estimate enteric CH_4 emissions have been developed using data sets of animals fed high-forage diets or mixed diets; few studies have fed high-concentrate diets typical of today's U.S. feedlots.

Table 5-E-1: Models Potentially Useful in Estimating Enteric CH_4 Emissions from Typical U.S. Ruminant Animals

Reference	Variable modeled	Inputs/Comments
Empirical Models		
IPCC (2006)	Enteric CH_4	No. of animals, animal species, animal type, emission factor for each animal type (Tier 2 CH_4 conversion factor; Ym)
Kriss (1930)	Enteric CH_4	Dry matter intake (DMI)
Axelsson (1949)	Enteric CH_4	DMI
Bratzler & Forbes (1940)	Enteric CH_4	Digested carbohydrate
Mills et al. (2003)	Enteric CH_4	Metabolizable energy (ME) intake, starch and acid detergent fiber (ADF) intake
Blaxter & Clapperton (1965)	Enteric CH_4	Digestible energy (DE) (%) at maintenance intake, gross energy intake (GEI), feeding level (multiple of maintenance)
Moe & Tyrrell (1979)	Enteric CH_4	Digestible soluble carbohydrates, digestible hemicellulose, digestible cellulose
Holter & Young (1992)	Enteric CH_4	Digestible soluble carbohydrates, cellulose, hemicellulose, fat intake
Yan et al. (2009)	Enteric CH_4	Digestible energy, silage, and total DMI, silage, and diet ADF
Ellis et al. (2007)	Enteric CH_4	Metabolizable energy intake, ADF, lignin intake
Ellis et al. (2009)	Enteric CH_4	Metabolizable energy intake, cellulose, hemicellulose, and fat intake; non-fiber carbohydrate, neutral detergent fiber (NDF), and DMI
Mills et al. (2001)	Enteric CH_4	DMI
Holos (Little et al., 2008)	Enteric CH_4, manure CH_4	Based on IPCC (2006)
CNCPS (2010)	Enteric CH_4, DMI, nutrient excretion, urine nitrogen excretion;	Uses equation of Mills et al. (2003) for dairy and Ellis et al. (2007) for beef. Animal characteristics, diet nutrient composition, feed protein fractions, animal performance, animal management, in situ degradability of feeds
Integrated Farm System Model (Rotz et al., 2011b)	Enteric CH_4, nutrient excretion, urine nitrogen, DMI, manure NH_3, CH_4, and N_2O	Uses the Mits3 equation of Mills et al. (2003) for enteric CH_4, IPCC (2006) for manure CH_4, and either DAYCENT (Chianese et al., 2009d) or IPCC (2006) for manure N_2O
Phetteplace et al. (2001)	Enteric CH_4, manure CH_4	Animal class, animal age and body weight, quantity of meat/mile produced, feed type, feed intake, manure management
Process-based Models		

Reference	Variable modeled	Inputs/Comments
Kebreab et al., (2004; 2009)	Enteric CH_4, nutrient excretion	DMI, NDF, degradable NDF, total starch, degradable starch, soluble sugars in diet, diet nitrogen, NHx-N in diet, indigestible protein, rate of degradation of starch, and protein
COWPOLL (Dijkstra et al., 1992; Mills et al., 2003; Bannink et al., 2006; Kebreab et al., 2008)	Enteric CH_4	DMI, NDF, degradable NDF, total starch, degradable starch, soluble sugars in diet, diet nitrogen, NHx-N in diet, indigestible protein, rate of degradation of starch, and protein
MOLLY (Baldwin, 1995)	Enteric CH_4	Similar to COWPOLL

Prediction models for enteric emissions. The following is a brief summary of the models evaluated and their strengths and limitations.

Simple Regression Model Based on Digestible Energy. Blaxter and Clapperton (1965)developed a simple regression equation to estimate enteric CH_4 based on digestible energy, feed intake as a percentage of maintenance and GEI. The data set used to create this empirical model was composed mostly of data from sheep fed low-concentrate diets in respiration chambers, which may account for its limited accuracy in predicting CH_4 emissions across ruminant diets (Johnson et al., 1991).

Empirical Model. Moe and Tyrrell (1979) developed an empirical model to estimate enteric CH_4 emission from dairy cows based on diet composition. This empirical model was developed with high-forage diets in dairy cows fed in respiration chambers; its use for estimating beef cattle enteric emissions is therefore limited.

Regression Model. Yan et al. (2000) developed regression equations to predict enteric CH_4 emissions from beef and dairy cattle fed diets based on grass silage. Concentrates represented from 0 to 81.5 percent of the DMI, with a mean of 46.7 percent of diet DMI. When corrected to equal feed intakes, animal body weight had no effect on enteric CH_4 emissions. (Yan et al., 2000) validated their equations using data from the literature, mostly dairy studies with all diets based on grass silage.

Regression Equations. Ellis et al. (2009) developed regression equations to estimate enteric CH_4 production from beef cattle based on studies in which cattle were fed high-concentrate or moderate-concentrate (50 percent) diets. These equations were compared with 14 equations developed earlier by Ellis et al. (2007), seven developed by Mills et al. (2003), the Blaxter and Clapperton (1965) equation, and the Moe and Tyrrell (1979) equation. The mean enteric CH_4 production (MJ day^{-1} and percent of GEI) in all 12 of the studies was greater than values noted more recently (Hales et al., 2012), possibly because of differences in dietary grain content and fat supplementation. However, some of the Ellis (2007; 2009) equations estimated CH_4 emissions similar to those reported by Todd et al. (2014a; 2014b) in open lot feedlots.

The linear model with the lowest residual mean square prediction error (RMSPE) was Equation 5-E-1 as follows:

Equation 5-E-1: Linear Model with the Lowest RMSPE

$$\mathbf{CH_4 = 2.72 + (0.0937 \times ME\ intake) + (4.31 \times CELL) - (6.49 \times HC) - (7.44 \times Fat)}$$

Where:

CH_4 = Methane per day (MJ day^{-1})

ME intake = ME intake in (MJ day^{-1})

CELL = Cellulose intake (kg day^{-1})

HC = Hemicellulose intake (kg day^{-1})

Fat = Fat intake (kg day^{-1})

A possible advantage to using this equation, compared with other empirical equations, is that the variables required for the calculations can be readily obtained with some training in nutrition. Another is that the independent variables in the model (energy, fiber, and fat intake) are the primary differences that would occur in various beef and dairy cattle diets. However, a major concern with their use for finishing cattle is that a number of the studies used to develop the equations were high-forage diets and/or did not use either supplemental fat or monensin in the diet. As previously noted, when compared with emissions from cattle fed typical finishing diets based on steam-flaked corn (SFC) or dry-rolled corn (DRC), this equation greatly overestimated CH_4 emissions (Hales et al., 2012). Linear equations using nutrient ratios (starch:NDF, etc.) were also developed, but all had greater RMSPE than the previous equation (Ellis et al., 2009). Nonlinear equations were also developed. Despite being more biologically defendable, the nonlinear equations all had greater RMSPE than the linear equation.

In a later study, Yan et al. (2009) developed additional equations using a database of 108 measurements for beef steers of varied breeding in respiration chambers and fed diets that ranged from 100 to 30 percent roughage. They also compared a number of equations developed elsewhere. Equations were "validated" using one-third of the original data set. Emissions were highly correlated to live body weight, DMI, and GEI, but live body weight was a poor predictor of enteric CH_4 emissions. The ability of a number of equations to predict enteric CH_4 measured in the study was varied (eight percent overpredicted, to 33 percent underpredicted). The poorest results were with four linear equations developed by Ellis et al. (2007) that used DMI, MEI, and/or forage intake as independent variables. They attributed the poor response to the fact that a good portion of the data for Ellis et al. (2007) was from grazing animals using the SF_6 technique, which would not include CH_4 from the lower gut. The Blaxter and Clapperton (1965) equation did a respectable job (93 percent of actual with R^2 = 0.69; mean prediction error = 0.12; and 63 percent of means square prediction error due to random effects, and 29 percent due to a mean bias).

Empirical and Mechanistic Model. The IFSM Model (and its subset DairyGEM) (Rotz et al., 2005; Chianese et al., 2009b; 2009c; 2009a; 2009d) is a combination empirical and mechanistic model of whole farm nutrient management. The submodel to estimate enteric CH_4 emissions from beef or dairy cattle uses the Mits3 equation of Mills et al. (2003). Ellis et al. (2007) reported that the Mills et al. (2003) equations were poor at predicting CH_4 from beef cattle, probably because they were developed from dairy data. In fact, one equation that worked well with dairy cows actually predicted negative CH_4 emissions from beef cattle fed high-concentrate, low-forage diets. Thus, the current IFSM may not be appropriate to estimate enteric CH_4 emissions from beef cattle, especially feedlot cattle.

Mechanistic Models. MOLLY (Baldwin et al., 1987; Baldwin, 1995) is a mechanistic model that estimates ruminal CH_4 production based on a hydrogen balance within the rumen. Input parameters to the model are daily DMI, chemical composition of the diet, solubility of protein and starch, degradability, ruminal passage rates, ruminal volume, and ruminal pH. COWPOLL (Dijkstra et al., 1992; Mills et al., 2001) is another mechanistic model. Input parameters to the model are similar to MOLLY. MOLLY and COWPOLL both use an H-balance to estimate enteric CH_4 production. However, they use different VFA stoichiometry submodels. Both models require significant inputs that are probably beyond the scope of typical producers. However, they are excellent research tools.

The Cornell Net Carbohydrate and Protein System model (CNCPS, 2010) calculates nutrient requirements, nutrient inputs, animal production (weight gain and/or milk production), and nutrient excretion in beef and dairy cattle. It recently added a submodel (VanAmburgh et al., 2010) to calculate enteric CH_4 emissions. The submodel uses an equation of Mills et al. (2003) to estimate enteric emissions from dairy cows and an equation of Ellis et al. (2007) to estimate enteric emissions from beef cattle. At present, to our knowledge there are no comparisons or independent validations of the new submodels that have been published, and the extent to which the model is responsive to mitigation strategies is unclear.

Comparative Analyses using Independent Data Sets. Several studies have attempted to evaluate the predictive ability of enteric CH_4 models by using an independent data set. Benchaar et al. (1998) compared two mechanistic (Baldwin et al., 1987; Dijkstra et al., 1992; Baldwin, 1995); and two linear (Blaxter and Clapperton, 1965; Moe and Tyrrell, 1979) models with a data set of 32 diets from 13 publications in the literature. They noted that the mechanistic models were better predictors than the regression equations. The linear regression models could only explain 42 to 57 percent of the variation in predicted values, whereas the mechanistic models explained more than 70 percent of the variation. The model of Dijkstra et al. (1992) tended to underestimate actual CH_4 production (mean error = 0.30 Mcal day^{-1}), with the error being greater at higher CH_4 productions. The model of Baldwin (Baldwin et al., 1987; Baldwin, 1995) overestimated CH_4 production by about 0.93 Mcal day^{-1}, primarily due to a high intercept. The equations of Moe and Tyrrell (1979) and Blaxter and Clapperton (1965) tended to overestimate CH_4 production, especially at low production rates.

Comparative Analysis/Lactating and Nonlactating Cows. Wilkerson et al. (1995) compared several published equations (Kriss, 1930; Bratzler and Forbes, 1940; Axelsson, 1949; Blaxter and Clapperton, 1965; Moe and Tyrrell, 1979; Holter and Young, 1992) for their ability to predict enteric CH_4 production from lactating and nonlactating Holstein cows. In general, equations that were based on total DMI or on intake of digested cellulose, hemicellulose, and nonfiber carbohydrates, provided the highest correlation and lowest errors of prediction. Prediction equations that used a quadratic function of DMI were poor at predicting enteric CH_4. In general, the equations predicted emissions from nonlactating cows more accurately than from lactating cows.

Comparative Analysis Linear Models. Kebreab et al. (2006) compared two linear models (Moe and Tyrrell, 1979; Mills et al., 2003), a nonlinear model (Mills et al., 2003), the IPCC Tier 1 and Tier 2 models (IPCC, 1997), and a dynamic mechanistic model (Kebreab et al., 2004) using data from studies conducted in North America. They recommended that the linear models be used when there is limited information on nutrient intake and when the expected emissions are within the range of data from which the model was developed. The nonlinear model of Mills et al. (2003) could be used for extrapolating beyond the range of data used to develop the equation, but the mechanistic model was recommended for evaluation of mitigation options. The IPCC Tier 1 model was found to be adequate for general inventory purposes. The predictive ability of the Tier 2 model, while most useful, was limited.

Comparative Analysis Mechanistic Models. Kebreab et al. (2008) also compared two mechanistic models, MOLLY (Baldwin et al., 1987; Baldwin, 1995) and COWPOLL (Dijkstra et al., 1992; Mills et al., 2001; Bannink et al., 2006), to the IPCC Tier 2 (2006) and linear equation of Moe and Tyrrell (1979). Using a beef cattle data set, MOLLY and IPCC tended to be more accurate than the other models, although MOLLY was more precise. MOLLY and IPCC Tier 2 had minimal mean bias, whereas COWPOLL and the Moe and Tyrrell (1979) equation greatly overpredicted average emissions. COWPOLL, which is based on the enteric CH_4 prediction equations of Mills et al. (2001) and the updated rumen stoichiometry for lactating cows (Bannink et al., 2006), had the poorest ability to predict enteric CH_4 emission from feedlot cattle and tended to overpredict CH_4 emissions (MJ day^{-1}) by as much as 50 percent. Although on average MOLLY and IPCC Tier 2 (2006) gave predicted values similar to measured values, there was a large variability in individual animals, with errors of 75 percent or greater. The large variability in predicted values indicates that there can be large animal-to-animal variation in enteric CH_4 production, even when animals are fed the same diets at similar feed intakes.

Comparative Analysis/Feedlots. McGinn et al. (2008) compared measured (using bLS model) CH_4 emissions (enteric plus pen surface) from feedlots in Australia and Canada with estimates using the IPCC Tier 1, IPCC Tier 2, Blaxter and Clapperton (1965), and Moe and Tyrrell (1979) equations. The Tier 2 method underestimated CH_4 at both locations. Estimates using the IPCC Tier 1 methods were close to measured values in Australia; however, Tier 1 underestimated values for the Canada feedlot. Estimates made using the Blaxter and Clapperton (1965) and Moe and Tyrrell (1979) equations were close to measured values in Canada, but overestimated values in Australia. Methane emissions had a significant diel pattern indicating that short-term measurement of CH_4 emissions at feedlots may overestimate or underestimate daily emissions.

Comparative Analysis of Stoichiometric Models. Alemu et al. (2011) compared enteric CH_4 emissions from dairy cows using a variety of stoichiometric models of ruminal fermentation (Murphy et al., 1982; Bannink et al., 2006; Sveinbjornsson et al., 2006; Nozière et al., 2010), and noted that mechanistic models such as Bannink et al. (2006) are more accurate for predicting enteric CH_4 from dairy cows than the IPCC Tier 2 (2006) method. However, these models required a considerable quantity of data regarding the animals and their diet.

Comparative Analysis Measurement Data and Models. Tomkins et al. (2011) measured enteric CH_4 emissions of steers on pasture using a micrometeorological method and respiration chambers. Emissions estimated using an Ellis (2009) equation (CH_4, MJ day^{-1} = 3.272 +0.736 (DMI, kg day^{-1})) were similar (112.7 g day^{-1}) to measured emissions. Estimates using the equations of Kurihara et al. (1999) as modified by Hunter (2007) (109.1 g day^{-1}), Yan et al. (2009) (105.6 g day^{-1}), and Charmley et al. (2008) (2008:NABCEMS; 100.2 g day^{-1}) were slightly lower, but not as low as the IPCC (2006) model (82.7 g day^{-1}).

Comparative Analyses/Models. Legesse et al. (2011) compared enteric CH_4 emission estimates using MOLLY, COWPOLL, IPCC Tier 2, and one equation of Ellis et al. (2007) under various Canadian beef cow-calf management systems. Differences among the models (26 to 35 percent) were much greater than differences among management systems (three to five percent). The authors suggested that these differences limited the model's utility in predicting CH_4 emission from beef cow systems.

Evaluation of Models. Yan et al. (2000; 2009) noted that CH_4 production (percent of GEI or digestible energy) decreased with increasing DMI (as multiples of maintenance) and with increasing forage in the diet. Thus, they suggested that models that do not consider feeding level will underpredict CH_4 at low planes of nutrition and overpredict enteric CH_4 at high levels of feeding. Similarly, Kebreab et al. (2006) noted that linear models tend to give unrealistically high emission values when DMI increases, whereas nonlinear models gave values approaching the theoretical maximum emission, which is biologically reasonable.

Although several equations of Ellis et al. (2009) appeared to be good predictors of enteric CH_4 losses from feedlot cattle based on Canadian studies, when compared with data from cattle fed a typical corn-based finishing diet (Hales et al., 2012) most tended to greatly overestimate enteric losses. At the present time, the IPCC Tier 2 model with some modifications may be the most useful for prediction of enteric emissions from feedlot beef cattle.

Chapter 5 References

Aarnink, A.J.A., and M.W.A. Verstegen. 2007. Nutrition, key factor to reduce environmental load from pig production. *Livestock Science*, 109(1–3):194-203.

Adviento-Borbe, M.A.A., E.F. Wheeler, N.E. Brown, P.A. Topper, et al. 2010. Ammonia and greenhouse gas flux from manure in freestall barn with dairy cows on precision fed rations. *Transactions of the ASABE*, 53:1251-1266.

Agnew, J., C. Laguë, J. Schoenau, and R. Farrell. 2010. Greenhouse gas emissions measured 24 hours after surface and subsurface application of different manure types. *Transactions of the ASABE*, 53(5):1689-1701.

Aguerre, M.J., M.A. Wattiaux, J.M. Powell, G.A. Broderick, et al. 2011. Effect of forage-to-concentrate ratio in dairy cow diets on emission of methane, carbon dioxide, and ammonia, lactation performance, and manure excretion. *Journal of Dairy Science*, 94(6):3081-3093.

Alemu, A.W., J. Dijkstra, A. Bannink, J. France, et al. 2011. Rumen stoichiometric models and their contribution and challenges in predicting enteric methane production. *Animal Feed Science and Technology*, 166-167(0):761-778.

Amon, B., T. Amon, J. Boxberger, and C. Alt. 2001. Emissions of NH3, N2O and CH4 from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). *Nutrient Cycling in Agroecosystems*, 60(1):103-113.

Anderson, R.C., and M.A. Rasmussen. 1998. Use of a novel nitrotoxin-metabolizing bacterium to reduce ruminal methane production. *Bioresource Technology*, 64(2):89-95.

Anderson, R.C., T.R. Callaway, J.A.S. Van Kessel, Y.S. Jung, et al. 2003. Effect of select nitrocompounds on ruminal fermentation; an initial look at their potential to reduce economic and environmental costs associated with ruminal methanogenesis. *Bioresource Technology*, 90(1):59-63.

Applegate, T., W. Powers, R. Angel, and D. Hoehler. 2008. Effect of Amino Acid Formulation and Amino Acid Supplementation on Performance and Nitrogen Excretion in Turkey Toms. *Poultry Science*, 87(3):514-520.

Archibeque, S.L., D.N. Miller, H.C. Freetly, and C.L. Ferrell. 2006. Feeding high-moisture corn instead of dry-rolled corn reduces odorous compound production in manure of finishing beef cattle without decreasing performance. *Journal of Animal Science*, 84(7):1767-1777.

Arriaga, H., G. Salcedo, L. Martínez-Suller, S. Calsamiglia, et al. 2010. Effect of dietary crude protein modification on ammonia and nitrous oxide concentration on a tie-stall dairy barn floor. *Journal of Dairy Science*, 93(7):3158-3165.

ASABE. 2005. Manure Production and Characteristics, ASABE Standard D384.2 MAR2005 (R2010). St. Joseph, MI: American Society of Agricultural and Biological Engineers.

Atakora, J.K.A., S. Möhn, and R.O. Ball. 2003. Low protein diets maintain performance and reduce greenhouse gas production in finisher pigs. Proceedings of the 2003 Banff Pork Seminar, Alberta, Canada.

Atakora, J.K.A., S. Möhn, and R.O. Ball. 2004. Effects of dietary protein reduction on greenhouse gas emission from pigs. Proceedings of the 2004 Banff Pork Seminar, Alberta, Canada.

Australian Greenhouse Office. 2007. *National greenhouse gas inventory 2005*. Canberra, Australia: Dept. of the Environment and Water Resources.

Axelsson, J. 1949. The amount of produced methane energy in the European metabolic experiments with adult cattle. *Annals of the Royal Agricultural College of Sweden*, 16:404-419.

Baker, C. 1969. *Temperature dependence of self-diffusion coefficients for gaseous ammonia*. Washington D.C.: Lewis Research Center, National Aeronautics and Space Administration.

Baldwin, R.L., J.H.M. Thornley, and D.E. Beever. 1987. Metabolism of the lactating cow: II. Digestive elements of a mechanistic model. *Journal of Dairy Research*, 54(01):107-131.

Baldwin, R.L. 1995. *Modeling ruminant Digestion and Metabolism*. London, UK: Chapman & Hall.

Ball, R.O., and S. Möhn. 2003. Feeding strategies to reduce greenhouse gas emissions from pigs. Proceedings of the 2003 Banff Pork Seminar, Alberta, Canada.

Bannink, A., J. Kogut, J. Dijkstra, J. France, et al. 2006. Estimation of the stoichiometry of volatile fatty acid production in the rumen of lactating cows. *Journal of Theoretical Biology*, 238(1):36-51.

Beauchemin, K., M. Kreuzer, F. O'Mara, and T. McAllister. 2008. Nutritional management for enteric methane abatement: a review. *Australian Journal of Experimental Agriculture*, 48:21-27.

Beauchemin, K.A., and S.M. McGinn. 2005. Methane emissions from feedlot cattle fed barley or corn diets. *Journal of Animal Science*, 83(3):653-661.

Beauchemin, K.A., and S.M. McGinn. 2006a. Methane emissions from beef cattle: Effects of fumaric acid, essential oil, and canola oil. *Journal of Animal Science*, 84(6):1489-1496.

Beauchemin, K.A., and S.M. McGinn. 2006b. Enteric methane emissions from growing beef cattle as affected by diet and level of intake. *Canadian Journal of Animal Science*, 86(3):401-408.

Beauchemin, K.A., H. Henry Janzen, S.M. Little, T.A. McAllister, et al. 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agricultural Systems*, 103(6):371-379.

Benchaar, C., J. Rivest, C. Pomar, and J. Chiquette. 1998. Prediction of methane production from dairy cows using existing mechanistic models and regression equations. *Journal of Animal Science*, 76(2):617-627.

Benchaar, C., C. Pomar, and J. Chiquette. 2001. Evaluation of dietary strategies to reduce methane production in ruminants: A modeling approach. *Canadian Journal of Animal Science*, 81:563-574.

Berger, L.L., and N.R. Merchen. 1995. Influence of protein level on intake of feedlot cattle – Role of ruminal ammonia supply. Symposium. Intake of Feedlot Cattle. Oklahoma State Univ. July, 1995:942:272-280.

Bhatta, R., K. Tajima, N. Takusari, K. Higuchi, et al. 2007. Comparison of in vivo and in vitro techniques for methane production from ruminant diets. *Asian-Australasian Journal of Animal Sciences*, 20:1049-1056.

Bhatti, J.S., R. Lal, M.A. Price, and M.J. Apps, (eds.). 2005. *Climate Change and Managed Ecosystems*. Boca Raton, FL: CRC Press.

Bjorneberg, D.L., A.B. Leytem, D.T. Westermann, P.R. Griffiths, et al. 2009. Measurement of atmospheric ammonia, methane, and nitrous oxide at a concentrated dairy production facility in southern Idaho using open-path FTIR spectrometry. *Transactions of the ASABE*, 52(5):1749-1756.

Blaxter, K.L., and F.W. Wainman. 1964. The utilization of the energy of different rations by sheep and cattle for maintenance and for fattening. *Journal of Agricultural Science*, 63(01):113-128.

Blaxter, K.L., and J.L. Clapperton. 1965. Prediction of the amount of methane produced by ruminants. *British Journal of Nutrition*, 19:511-522.

Bluteau, C.V., D.I. Massé, and R. Leduc. 2009. Ammonia emission rates from dairy livestock buildings in Eastern Canada. *Biosystems Engineering*, 103(4):480-488.

Boadi, D.A., K.M. Wittenberg, and W. McCaughey. 2002. Effects of grain supplementation on methane production of grazing steers using the sulphur (SF6) tracer gas technique. *Canadian Journal of Animal Science*, 82(2):151-157.

Branine, M.E., and D.E. Johnson. 1990. Level of intake effects on ruminant methane loss across a wide range of diets. *Journal of Animal Science*, 68 (Suppl. 1):509-510.

Bratzler, J.W., and E.B. Forbes. 1940. The estimation of methane production by cattle. *Journal of Nutrition*, 19:611-613.

Brooks, J.P., and M.R. McLaughlin. 2009. Antibiotic resistant bacterial profiles of anaerobic swine lagoon effluent. *Journal of Environmental Quality*, 38(6):2431-2437.

Cabrera, M.L., and S.C. Chiang. 1994. Water Content Effect on Denitrification and Ammonia Volatilization in Poultry Litter. *Soil Sci. Soc. Am. J.*, 58(3):811-816.

Cambardella, C.A., T.B. Moorman, and J.W. Singer. 2010. Soil nitrogen response to coupling cover crops with manure injection. *Nutrient Cycling in Agroecosystems*, 87(3):383-393.

Canh, T.T., M.W.A. Verstegen, A.J.A. Aarnink, and J.W. Schrama. 1997. Influence of dietary factors on nitrogen partitioning and composition of urine and feces of fattening pigs. *Journal of Animal Science*, 75:700-706.

Canh, T.T., A.J.A. Aarnink, Z. Mroz, A.W. Jongbloed, et al. 1998a. Influence of electrolyte balance and acidifying calcium salts in the diet of growing-finishing pigs on urinary pH, slurry pH and ammonia volatilisation from slurry. *Livest. Prod. Sci.*, 56:1-13.

Canh, T.T., A.J.A. Aarnink, J.B. Schutte, A.L. Sutton, et al. 1998b. Dietary protein affects nitrogen excretion and ammonia emission from slurry of growing finishing pigs. *Livestock Production Science*, 56:181-191.

Cantrell, K.B., T. Ducey, K.S. Ro, and P.G. Hunt. 2008a. Livestock waste-to-bioenergy generation opportunities. *Bioresource Technology*, 99(17):7941-7953.

Cantrell, K.B., K.S. Ro, and P.G. Hunt. 2008b. Thermal characterization of swine manure: Bioenergy feedstock potential.

Cantrell, K.B., K.C. Stone, P.G. Hunt, K.S. Ro, et al. 2009. Bioenergy from Coastal bermudagrass receiving subsurface drip irrigation with advance-treated swine wastewater. *Bioresource Technology*, 100(13):3285-3292.

Cantrell, K.B., P.G. Hunt, K.S. Ro, K.C. Stone, et al. 2010a. Thermogravimetric characterization of irrigated bermudagrass as a combustion feedstock. *Transactions of the ASABE*, 53(2):413-420.

Cantrell, K.B., J.H. Martin Ii, and K.S. Ro. 2010b. Application of thermogravimetric analysis for the proximate analysis of livestock wastes. *Journal of ASTM International*, 7(3).

Capper, J.L., R.A. Cady, and D.E. Bauman. 2009. The environmental impact of dairy production: 1944 compared with 2007. *Journal of Animal Science*, 87(6):2160-2167.

Carmean, B.R., K.A. Johnson, D.E. Johnson, and L.W. Johnson. 1992. Maintenance energy requirement of llama. *American Journal of Veterinary Reserach*, 53:1696-1698.

Carr, L.E., F.W. Wheaton, and L.W. Douglass. 1990. Empirical models to determine ammonia concentrations from broiler chicken litter. *Trans. ASAE*, 33:1337-1342.

Cassel, T., L. Ashbaugh, R. Flocchini, and D. Meyer. 2005. Ammonia emission factors for open-lot dairies: Direct measurements and estimation by nitrogen intake. *Journal of the Air & Waste Management Association*, 55:826-833.

CDM. 2012. *Project and leakage emissions from anaerobic digesters. Ver. 01.0.0*: Clean Development Mechanism.

Charmley, E., M.L. Stephens, and P.M. Kennedy. 2008. Predicting livestock productivity and methane emissions in northern Australia: development of a bio-economic modeling approach. *Australian Journal of Experimental Agriculture*, 49:109-113.

Chastain, J.P., W.D. Lucas, J.E. Albrecht, J.C. Pardue, et al. 1998. *Solids and Nutrient Removal From Liquid Swine Manure Using a Screw Press Separator*, ASAE Paper No. 98-4110. St. Joseph, MI: American Society of Agricultural Engineers.

Chastain, J.P., M.B. Vanotti, and M.M. Wingfield. 2001. Effectiveness of Liquid-Solid Separation for Treatment of Flushed Dairy Manure: A Case Study. *Transactions of the American Society of Agricultural Engineers*, 17(3):343-354.

Chen, G.Q., L. Shao, Z.M. Chen, Z. Li, et al. 2011. Low-carbon assessment for ecological wastewater treatment by a constructed wetland in Beijing. *Ecological Engineering*, 37(4):622-628.

Chianese, D.S., C.A. Rotz, and T.L. Richard. 2009a. Simulation of carbon dioxide emissions from dairy farms to assess greenhouse gas reduction strategies. *Trans. ASABE*, 52:1301-1312.

Chianese, D.S., C.A. Rotz, and T.L. Richard. 2009b. Whole-farm greenhouse gas emissions: A review with application to a Pennsylvania dairy farm. *Applied Engineering in Agriculture*, 25:431-442.

Chianese, D.S., C.A. Rotz, and T.L. Richard. 2009c. Simulation of methane emissions from dairy farms to assess greenhouse gas reduction strategies. *Trans. ASABE*, 52:1313-1323.

Chianese, D.S., C.A. Rotz, and T.L. Richard. 2009d. Simulation of nitrous oxide emissions from dairy farms to assess greenhouse gas reduction strategies. *Trans. ASABE*, 52:1325-1335.

Chiumenti, R., L. Donatoni, and S. Guercini. 1987. Liquid / Solid Separation Tests on Beef Cattle Manure. Proceedings of the Seminar of the 2nd Technical Section of the C.I.G.R., Ubrana Champaign, IL, June 22-26, 1987.

Clark, O., S. Moehn, J. Edeogu, J. Price, et al. 2005. Manipulation of dietary protein and nonstarch polysaccharide to control swine manure emissions. *Journal of Environmental Quality*, 34:1461-1466.

CNCPS. *Cornell Net Carbohydrate and Protein System*. Retrieved from http://www.cncps.cornell.edu.

Cole, N.A., and J.E. McCroskey. 1975. Effects of Hemiacetal of Chloral and Starch on the Performance of Beef Steers. *Journal of Animal Science*, 41(6):1735-1741.

Cole, N.A., P.J. Defoor, M.L. Galyean, G.C. Duff, et al. 2006. Effects of phase feeding of crude protein on performance, carcass characteristics, serum urea nitrogen concentrations and manure nitrogen in finishing bee steers. *Journal of Animal Science*, 84:3421-3432.

Cole, N.A., R.W. Todd, D.B. Parker, and M. Rhoades. 2007b. Challenges in using flux chambers to measure ammonia emissions form simulated feedlot pen surfaces and retention ponds. Proceedings of the International Symposium On Air Quality and Waste Management for Agriculture, Sept. 16-19, 2007, Broomfield, CO.

Cole, N.A., A.M. Mason, R.W. Todd, and D.B. Parker. 2009a. Effects of urine application on chemistry of feedlot pen surfaces. *Journal of Animal Science*, 87:E Suppl. 2:148 (Abstract).

Cole, N.A., A.M. Mason, R.W. Todd, M. Rhoades, et al. 2009b. Chemical Composition of Pen Surface Layers of Beef Cattle Feedyards. *The Professional Animal Scientist*, 25(5):541-552.

Converse, J.C., R.G. Koegel, and R.J. Straub. 1999. *Nutrient and Solids Separation of Dairy and Swine Manure Using a Screw Press Separator*, ASAE Paper No. 99-4050. St. Joseph, MI: American Society of Agricultural Engineers.

Cooprider, K.L., F.M. Mitloehner, T.R. Famula, E. Kebreab, et al. 2011. Feedlot efficiency implications on greenhouse gas emission and sustainability. *Journal of Animal Science*, (in press).

Cornell University Department of Animal Science. 2010. CNCPS.

Corrigan, M.E., T.J. Klopfenstein, G.E. Erickson, N.F. Meyer, et al. 2009. Effects of level of condensed distiller's solubles in corn dried distillers grains on intake, daily weight gain, and digestibility in growing steers fed forage diets. *Journal of Animal Science*, 87:4073-4081.

Cottle, D.J., J.V. Nolan, and S.G. Wiedemann. 2011. Ruminant enteric methane mitigation: a review. *Animal Production Science*, 51(6):491-514.

Coufal, C., C. Chavez, P. Niemeyer, and J. Carey. 2006. Nitrogen emissions from broilers measured by mass balance over eighteen consecutive flocks. *Poultry Science*, 85(3):384-391.

Crutzen, P.J., I. Aselmann, and W. Seiler. 1986. Methane production by domestic animals, wild ruminants, other herbivorous fauna, and humans. *Tellus B*, 38B(3-4):271-284.

Davies, P.R. 2011. Intensive swine production and pork safety. *Foodborne Pathogens and Disease*, 8(2):189-201.

Delfino, J., G.W. Mathison, and M.W. Smith. 1988. Effect of lasalocid on feedlot performance and energy partitioning in cattle. *Journal of Animal Science*, 66:236-241.

Delmore, R.J., J.M. Hodgen, and B.J. Johnson. 2010. Perspectives on the application of zilpaterol hydrochloride in the United States beef industry. *Journal of Animal Science*, 88:2825-2828.

DeRamus, H., T. Clement, D. Giampola, and P. Dickison. 2003. Methane emissions of beef cattle on forages: efficiency of grazing management systems. *Journal of Environmental Quality*, 21:269-277.

Dibner, J., and J. Richards. 2005. Antibiotic growth promoters in agriculture: history and mode of action. *Poultry Science*, 84(4):634-643.

Dijkstra, J., H.D.S.C. Neal, D.E. Beever, and J. France. 1992. Simulation of nutrient digestion, absorption and outflow in the rumen: model description. *Journal of Nutrition*, 122(1992):2239-2256.

Dijkstra, J., E. Kebreab, J.A.N. Mills, W.F. Pellikaan, et al. 2007. Predicting the profile of nutrients available for absorption: from nutrient requirement to animal response and environmental impact. *Animal*, 1(1):99-111.

Dini, Y., J. Gere, C. Briano, M. Manetti, et al. 2012. Methane emission and milk production of dairy cows grazing pastures righ in legumes or rich in grasses in Uruguay. *Animals*, 2:288-300.

Dodla, S.K., J.J. Wang, R.D. DeLaune, and R.L. Cook. 2008. Denitrification potential and its relation to organic carbon quality in three coastal wetland soils. *Science of the Total Environment*, 407(1):471-480.

Dong, R., Y. Zhang, L.L. Christianson, T.L. Funk, et al. 2009. Product distribution and implication of hydrothermal conversion of swine manure at low Temperatures. *Transactions of the ASABE*, 52(4):1239-1248.

Eckard, R.J., C. Grainger, and C.A.M. de Klein. 2010. Options for the abatement of methane and nitrous oxide from ruminant production: A review. 130(1):47-56.

Elam, N.A., J.T. Vasconcelos, G. Hilton, D.L. VanOverbeke, et al. 2009. Effect of zilpaterol hydrochloride duration of feeding on performance and carcass characteristics of feedlot cattle. *Journal of Animal Science*, 87:2133-2141.

Elgood, Z., W.D. Robertson, S.L. Schiff, and R. Elgood. 2010. Nitrate removal and greenhouse gas production in a stream-bed denitrifying bioreactor. *Ecological Engineering*, 36(11):1575-1580.

Elliot, H.A., and N.E. Collins. 1982. Factors affecting ammonia release in broiler houses. *Trans. ASAE*, 25:413-418.

Ellis, J.L., E. Kebreab, N.E. Odongo, B.W. McBride, et al. 2007. Prediction of Methane Production from Dairy and Beef Cattle. *Journal of Dairy Science*, 90(7):3456-3466.

Ellis, J.L., E. Kebreab, N.E. Odongo, K. Beauchemin, et al. 2009. Modeling methane production from beef cattle using linear and nonlinear approaches. *Journal of Animal Science*, 87(4):1334-1345.

Ellis, S., J. Webb, T. Misselbrook, and D. Chadwick. 2001. Emission of ammonia (NH3), nitrous oxide (N2O) and methane (CH4) from a dairy hardstanding in the UK. *Nutrient Cycling in Agroecosystems*, 60(1):115-122.

Ewan, R.C. 1989. Predicting the energy utilization of diets and feed ingredients by pigs. In *Energy Metabolism, European Association of Animal Production Bulletin No. 43*, Y. v. d. Honing and W. H. Close (eds.). Pudoc Wageningen, Netherlands.

Faubert, P., P. Tiiva, Å. Rinnan, S. Räty, et al. 2010. Effect of vegetation removal and water table drawdown on the non-methane biogenic volatile organic compound emissions in boreal peatland microcosms. *Atmospheric Environment*, 44(35):4432-4439.
Faulwetter, J.L., V. Gagnon, C. Sundberg, F. Chazarenc, et al. 2009. Microbial processes influencing performance of treatment wetlands: A review. *Ecological Engineering*, 35(6):987-1004.
Ferguson, N., R. Gates, J. Taraba, A. Cantor, et al. 1998a. The effect of dietary crude protein on growth, ammonia concentration, and litter composition in broilers. *Poultry Science*, 77(10):1481-1487.
Ferguson, N., R. Gates, J. Taraba, A. Cantor, et al. 1998b. The effect of dietary protein and phosphorus on ammonia concentration and litter composition in broilers. *Poultry Science*, 77(8):1085-1093.
Ferket, P., E. Heugten, T. Kempen, and R. Angel. 2002. Nutritional strategies to reduce environmental emission from non ruminants. *Journal of Animal Science*, 80(Suppl. 2):168-182.
Fernandes, L., E. McKyes, and L. Obidniak. 1988. Performance of a continuous belt microscreening unit for solid liquid separation of swine waste. *Transactions of the CSAE*, 30(1):151-155.
Fey, A., G. Benckiser, and J.C.G. Ottow. 1999. Emissions of nitrous oxide from a constructed wetland using a groundfilter and macrophytes in waste-water purification of a dairy farm. *Biology and Fertility of Soils*, 29(4):354-359.
Flesch, T.K., J.D. Wilson, L.A. Harper, and B.P. Crenna. 2005. Estimating gas emissions from a farm with an inverse-dispersion technique. *Atmospheric Environment*, 39(27):4863-4874.
Flesch, T.K., L.A. Harper, J.M. Powell, and J.D. Wilson. 2009. *Inverse-dispersion calculation of ammonia emissions from Wisconsin dairy farms*, 52: Transactions of the ASABE.
Flesch, T.K., R.L. Desjardins, and D. Worth. 2011. Fugitive methane emissions from an agricultural biodigester. *Biomass and Bioenergy*, 35(9):3927-3935.
Florin, N.H., A.R. Maddocks, S. Wood, and A.T. Harris. 2009. High-temperature thermal destruction of poultry derived wastes for energy recovery in Australia. *Waste Management*, 29(4):1399-1408.
Ford, M., and R. Fleming. 2002. *Mechnical Solid-Liquid Separation of Livestock Manure Literature Review*: University of Guelph.
Fournier. *Rotary Press*. Retrieved from http://www.rotary-press.com/.
Fowler, D., M. Coyle, C. Flechard, K. Hargreaves, et al. 2001. Advances in micrometeorological methods for the measurement and interpretation of gas and particle nitrogen fluxes. *Plant and Soil*, 228(1):117-129.
Freeman, C., M.A. Lock, S. Hughes, B. Reynolds, et al. 1997. Nitrous oxide emissions and the use of wetlands for water quality amelioration. *Environmental Science and Technology*, 31(8):2438-2440.
Freetly, H.C., and T.M. Brown-Brandl. 2013. Enteric methane production from beef cattle that vary in feed efficiency. *Journal of Animal Science*, 91(10):4826-4831.
Fukummoto, Y., T. Osada, D. Hanajima, and K. Haga. 2003. Patterns and quantities of NH_3, N_2O, and CH_4 emissions during swine manure composting without forced aeration- effect of compost pile scale. *Bioresource Technology*, 89:109-114.
Galbraith, J.K., G.W. Mathison, R.J. Hudson, T.A. McAllister, et al. 1998. Intake, digestibility, methane and heat production in bison, wapiti and white-tailed deer. *Canadian journal of animal science.*, 78(4):681-691.
Gao, F., and S.R. Yates. 1998. Simulation of enclosure-based methods for measuring gas emissions from soil to the atmosphere. *Journal of Geophysical Research*, 103(D20):26127-26136.
Gilbertson, C.B., and J.A. Nienaber. 1978. Separation of Coarse Solids from Beef Cattle Manure. *Transactions of the American Society of Agricultural Engineers*, 21(6):1185-1188.

Gleghorn, J.F., N.A. Elam, M.L. Galyean, G.C. Duff, et al. 2004. Effects of crude protein concentration and degradability on performance, carcass characteristics, and serum urea nitrogen concentrations in finishing beef steers. *Journal of Animal Science*, 82:2705-2717.

Glerum, J.C., G. Klomp, and H.R. Poelma. 1971. The Separation of Solid and Liquid Parts of Pig Slurry. Proceedings of the 1st International Symposium on Livestock Wastes, Columbus, OH, April 19-22, 1971.

Goopy, J.P., R. Woodgate, A. Donaldson, D.L. Robinson, et al. 2011. Validation of a short-term methane measurement using portable static chambers to estimate daily methane production in sheep. *Animal Feed Science and Technology*, 166-167(0):219-226.

Gould-Wells, D., and D.W. Williams. 2004. Biogas production from a covered lagoon digester and utilization in a microturbine.

Guan, H., K.M. Wittenberg, K.H. Ominski, and D.O. Krause. 2006. Efficacy of ionophores in cattle diets for mitigation of enteric methane. *Journal of Animal Science*, 84(7):1896-1906.

Hales, K.E., N.A. Cole, and J.C. MacDonald. 2012. Effects of corn processing method and dietary inclusion of wet distillers grain with solubles on energy metabolism and enteric methane emissions of finishing cattle. *Journal of Animal Science*, 90:3174-3185.

Hales, K.E., N.A. Cole, and J.C. MacDonald. 2013. Effects of increasing concentrations of wet distillers grains with solubles in steam-flaked corn-based diets on energy metabolism, carbon-nitrogen balance, and enteric methane emissions of cattle. *Journal of Animal Science*, 91:819-828.

Hales, K.E., T.M. Brown-Brandl, and H.C. Freetly. 2014. Effects of decreased dietary roughage concentration on energy metabolism and nutrient balance in finishing beef cattle. *Journal of Animal Science*, 92:264-271.

Hamilton, S.W., E.J. DePeters, J.A. McGarvey, J. Lathrop, et al. 2010. Greenhouse Gas, Animal Performance, and Bacterial Population Structure Responses to Dietary Monensin Fed to Dairy Cows. *Journal of Environmental Quality*, 39(1):106-114.

Hammer, D.A., (ed.) 1989. *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural.* Chelsea, MI: Lewis Publishers.

Hanni, S., J. DeRouchey, M. Tokach, R. Goodband, et al. 2007. The effects of dietary chicory and reduced nutrient diets on composition and odor of stored swine manure. *The Professional Animal Scientist*, 23:438-447.

Hansen, R.R., D.A. Nielsen, A. Schramm, L.P. Nielsen, et al. 2009. Greenhouse gas microbiology in wet and dry straw crust covering pig slurry. *Journal of Environmental Quality*, 38(3):1311-1319.

Hare, E., H.D. Norman, and J.R. Wright. 2006. Survival rate and productive herd life of dairy cattle in the United States. *Journal of Dairy Science*, 89(3713-3720).

Harper, L.A., O.T. Denmead, J.R. Freney, and F.M. Byers. 1999. Direct measurements of methane emissions from grazing and feedlot cattle. *Journal of Animal Science*, 77(6):1392-1401.

Harper, L.A. 2005. Ammonia: Measurement Issues. In *Micrometeorology in Agricultural Systems*, J. L. Hatfield and J. M. Baker (eds.). Madison, WI: American Society of Agronomy, Crop Science Society of America, Soil Science Society of America.

Harper, L.A., O.T. Denmead, and T.K. Flesch. 2011. Micrometeorological techniques for measurement of enteric greenhouse gas emissions. *Animal Feed Science and Technology*, 166-167(0):227-239.

Harrington, C., and M. Scholz. 2010. Assessment of pre-digested piggery wastewater treatment operations with surface flow integrated constructed wetland systems. *Bioresource Technology*, 101(20):7713-7723.

Harrington, R., and R. McInnes. 2009. Integrated Constructed Wetlands (ICW) for livestock wastewater management. *Bioresource Technology*, 100(22):5498-5505.

Hartung, J., and V. Phillips. 1994. Control of gaseous emissions from livestock buildings and manure stores. *Journal of Agricultural Engineering Research*, 57:173-189.

Hatfield, J.L., and R.L. Pfeiffer. 2005. Evaluation of technologies for ambient air monitoring at concentrated animal feeding operations.

Havenstein, G., P. Ferket, and M. Qureshi. 2003. Growth, livability, and feed conversion of 1957 versus 2001 broilers when fed representative 1957 and 2001 broiler diets. *Poultry Science*, 82(10):1500-1508.

Havenstein, G.B., P.R. Ferket, J.L. Grimes, M.A. Qureshi, et al. 2007. Comparison of the Performance of 1966- Versus 2003-Type Turkeys When Fed Representative 1966 and 2003 Turkey Diets: Growth Rate, Livability, and Feed Conversion. *Poultry Science*, 86(2):232-240.

Hayes, E., A. Leek, T. Curran, V. Dodd, et al. 2004. The influence of diet crude protein level on odour and ammonia emissions from finishing pig houses. *Bioresource Technology*, 91:309-315.

He, B., Y. Zhang, Y. Yin, T.L. Funk, et al. 2001. Effects of feedstock pH, initial CO addition, and total solids content on the thermochemical conversion process of swine manure. *Transactions of the American Society of Agricultural Engineers*, 44(3):697-701.

He, B.J., Y. Zhang, Y. Yin, T.L. Funk, et al. 2000. Operating temperature and retention time effects on the thermochemical conversion process of swine manure. *Transactions of the American Society of Agricultural Engineers*, 43(6):1821-1825.

Hegarty, R.S., J.P. Goopy, R.M. Herd, and B. McCorkell. 2007. Cattle selected for lower residual feed intake have reduced daily methane production. *Journal of Animal Science*, 85(6):1479-1486.

Hegg, R.O., R.E. Larson, and J.A. Moore. 1981. Mechanical Liquid-Solid Separation in Beef, Dairy, and Swine Waste Slurries. 24(1):0159-0163.

Hellebrand, H.J., and W.D. Kalk. 2000. Emissions caused by manure composting. *Agrartechnische Forschung*, 6(2):26-31.

Herschler, R.C., A.W. Olmsted, A.J. Edwards, R.L. Hale, et al. 1995. Production responses to various doses and ratios of estradiol benzoate and trenbolone acetate implants in steers and heifers. *Journal of Animal Science*, 73:2873-2881.

Hill, G.M., K.L. Richardson, and P.R. Utley. 1988. Feedlot performance and pregnancy inhibition of heifers treated with depot-formulated melengestrol acetate. *Journal of Animal Science*, 66:2435-2442.

Holmberg, R.D., D.T. Hill, T.J. Prince, and N.J. Van Dyke. 1983. Potential of Solid-Liquid Separation of Swine Wastes for Methane Production. *Transactions of the American Society of Agricultural Engineers*, 26(6):1803-1807.

Holter, J.B., and A.J. Young. 1992. Methane Prediction in Dry and Lactating Holstein Cows. *Journal of Dairy Science*, 75(8):2165-2175.

Howden, S.M., D.H. White, G.M. McKeon, J.C. Scanlan, et al. 1994. Methods for exploring management options to reduce greenhouse gas emissions from tropical grazing systems. *Climatic Change*, 27(1):49-70.

Hristov, A.N., M. Hanigan, A. Cole, R. Todd, et al. 2011. Review: Ammonia emissions from dairy farms and beef feedlots 1. *Canadian Journal of Animal Science*, 91(1):1-35.

Hristov, A.N. 2012. Historic, preEuropean settlement, and present-day contribution of wild ruminants to enteric methane emissions in the United States. *Journal of Animal Science*, 90:1371-1375.

Hunt, P.G., A.A. Szögi, F.J. Humenik, J.M. Rice, et al. 2002. Constructed wetlands for treatment of swine wastewater from an anaerobic lagoon. *Transactions of the ASAE*, 45(3):639-647.

Hunt, P.G., T.A. Matheny, and A.A. Szogi. 2003. Denitrification in constructed wetlands used for treatment of swine wastewater. *Journal of Environmental Quality*, 32(2):727-735.

Hunt, P.G., M.A. Matheny, and K.S. Ro. 2007. Nitrous oxide accumulation in soils from riparian buffers of a coastal plain watershed - carbon/nitrogen ratio control. *J. Environ. Qual.*, 36:1368-1376.

Hunter, R.A. 2007. Methane production by cattle in the tropics. *British Journal of Nutrition*, 98(03):657-657.

Hutchinson, G.L., and A.R. Mosier. 1981. Improved Soil Cover Method for Field Measurement of Nitrous Oxide Fluxes1. *Soil Sci. Soc. Am. J.*, 45(2):311-316.

Hwang, S., K. Jang, H. Jang, J. Song, et al. 2006. Factors affecting nitrous oxide production: A comparison of biological nitrogen removal processes with partial and complete nitrification. *Biodegradation*, 17(1):19-29.

Insam, H., and B. Wett. 2008. Control of GHG emission at the microbial community level. *Waste Management*, 28(4):699-706.

IPCC. 1997. *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme*. Bracknell, UK: Intergovernmental Panel on Climate Change. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html.

IPCC. 2000. *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories*: Intergovernmental Panel on Climate Change.

IPCC. 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme*. Japan: Intergovernmental Panel on Climate Change. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html.

Jarecki, M.K., T.B. Parkin, A.S.K. Chan, J.L. Hatfield, et al. 2008. Greenhouse gas emissions from two soils receiving nitrogen fertilizer and swine manure slurry. *Journal of Environmental Quality*, 37(4):1432-1438.

Jarecki, M.K., T.B. Parkin, A.S.K. Chan, T.C. Kaspar, et al. 2009. Cover crop effects on nitrous oxide emission from a manure-treated Mollisol. *Agriculture, Ecosystems and Environment*, 134(1-2):29-35.

Jin, Y., Z. Hu, and Z. Wen. 2009. Enhancing anaerobic digestibility and phosphorus recovery of dairy manure through microwave-based thermochemical pretreatment. *Water Research*, 43(14):3493-3502.

Johansson, A.E., Å. Kasimir Klemedtsson, L. Klemedtsson, and B.H. Svensson. 2003. Nitrous oxide exchanges with the atmosphere of a constructed wetland treating wastewater: Parameters and implications for emission factors. *Tellus, Series B: Chemical and Physical Meteorology*, 55(3):737-750.

Johnson, D.E. 1972. Effects of a Hemiacetal of Chloral and Starch on Methane Production and Energy Balance of Sheep Fed a Pelleted Diet. *Journal of Animal Science*, 35(5):1064-1068.

Johnson, D.E. 1974. Adaptational Responses in Nitrogen and Energy Balance of Lambs Fed a Methane Inhibitor. *Journal of Animal Science*, 38(1):154-157.

Johnson, D.E., T.M. Hill, B.R. Carmen, M.E. Branine, et al. 1991. New perspectives on ruminant methane emissions. In *Energy Metabolism of Farm Animals*, C. Wenk and M. Boessinger (eds.). Zurich, Switzerland: European Association for Animal Production.

Johnson, K., M. Huyler, H. Westberg, B. Lamb, et al. 1994. Measurement of methane emissions from ruminant livestock using a sulfur hexafluoride tracer technique. *Environmental Science & Technology*, 28(2):359-362.

Johnson, K.A., and D.E. Johnson. 1995. Methane emissions from cattle. *Journal of Animal Science*, 73(8):2483-2492.

Jones, F.M., F.A. Phillips, T. Naylor, and N.B. Mercer. 2011. Methane emissions from grazing Angus beef cows selected for divergent residual feed intake. *Animal Feed Science and Technology*, 166-167(0):302-307.

Jungbluth, T., E. Hartung, and G. Brose. 2001. . Greenhouse gas emissions from animal housing and manure stores. *Nutrient Cycling in Agroecosystems*, 60:133-145.

Kadlec, R.H., and R.L. Knight. 1996. *Treatment Wetlands*. Boca Ration, FL: Lewis Publishers.

Kebreab, E., J.A.N. Mills, L.A. Crompton, A. Bannink, et al. 2004. An integrated mathematical model to evaluate nutrient partition in dairy cattle between the animal and its environment. *Animal Feed Science and Technology*, 112(1-4):131-154.

Kebreab, E., K. Clark, C. Wagner-Riddle, and J. France. 2006. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. *Canadian journal of animal science.*, 86(2):135-137.

Kebreab, E., K.A. Johnson, S.L. Archibeque, D. Pape, et al. 2008. Model for estimating enteric methane emissions from United States dairy and feedlot cattle. *Journal of Animal Science*, 86(10):2738-2748.

Kebreab, E., J. Dijkstra, A. Bannink, and J. France. 2009. Recent advances in modeling nutrient utilization in ruminants. *Journal of Animal Science*, 87(14 suppl):E111-E122.

Kelliher, F.M., and H. Clark. 2010. Methane emissions from bison-An historic herd estimate for the North American Great Plains. *Agricultural and Forest Meteorology*, 150:473-477.

Kennedy, P.M., and E. Charmley. 2012. Methane yields from Brahman cattle fed tropical grasses and legumes. *Animal Production Science*, 52(4):225-239.

Kienbusch, M.R. 1986. *Measurement of gaseous emission rates from land surfaces using an emission-isolation flux chamber. User's guide*. PB-86-223161/XAB United StatesWed Feb 06 15:30:17 EST 2008NTIS, PC A04/MF A01.GRA; ERA-12-001567; EDB-86-185667English.

Kim, I., P. Ferket, W. Powers, H. Stein, et al. 2004. Effects of different dietary acidifier sources of calcium and phosphorus on ammonia, methane and odorant emission from growing-finishing pigs. . *Asian-Australasian Journal of Animal Sciences*, 17:1131-1138.

Kinsman, R., F.D. Sauer, H.A. Jackson, and M.S. Wolynetz. 1995. Methane and Carbon Dioxide Emissions from Dairy Cows in Full Lactation Monitored over a Six-Month Period. *Journal of Dairy Science*, 78(12):2760-2766.

Kirchgessner, M., M. Kreuzer, H. Muller, and W. Windisch. 1991. Release of methane and carbon dioxide by the pig. *Agricultural and Biological Research*, 44:103-133.

Klein, L., and A.D.G. Wright. 2006. Construction and operation of open-circuit methane chambers for small ruminants. *Australian journal of experimental agriculture.*, 46(10):1257-1262.

Klemedtsson, L., K. Von Arnold, P. Weslien, and P. Gundersen. 2005. Soil CN ratio as a scalar parameter to predict nitrous oxide emissions. *Global Change Biology*, 11(7):1142-1147.

Klieve, A.V., and R.S. Hegarty. 1999. Opportunities for biological control of ruminal methanogenesis: CSIRO.

Koelsch, R., and R. Stowell. 2005. *Ammonia Emissions Estimator*. Lincoln, NE: University of Nebraska. http://www.msue.msu.edu/objects/content_revision/download.cfm/revision_id.515204/workspace_id.27335/Forms%20for%20Estimating%20Swine%20and%20Dairy%20Emissions.pdf/.

Krehbiel, C.R., S.R. Rust, G. Zhang, and S.E. Gilliland. 2003. Bacterial direct-fed microbials in ruminant diets: Performance response and mode of action. *Journal of Animal Science*, 81(E. Suppl. 2):E120-E132.

Kreikemeier, W.M., and T.L. Mader. 2004. Effects of growth-promoting agents and season on yearling feedlot heifer performance. *Journal of Animal Science*, 82:2481-2488.

Kriss, M. 1930. Quantitative relations of the dry matter of the food consumed, the heat production, the gaseous outgo, and the insensible loss in body weight of cattle. *Journal of Agricultural Research*, 40:283-295.

KSU. 2012. *Focus on Feedlots*. Retrieved December 20 from http://www.asi.ksu.edu/p.aspx?tabid=302.

Kurihara, M., T. Magner, R.A. Hunter, and G.J. McCrabb. 1999. Methane production and energy partition of cattle in the tropics. *British Journal of Nutrition*, 81(03):227-234.

Kurup, R. 2003. Performance of a residential scale plug flow anaerobic reactor for domestic organic waste treatment and biogas generation. In *ORBIT 2003 Organic Recovery and BIological Treatment Proceedings of the Fourth International Conference of ORBIT Association on*

Biological Processing of Organics: Advances for a Sustainable Society. Perth, Western Australia.

Lassey, K.R. 2007. Livestock methane emission: From the individual grazing animal through national inventories to the global methane cycle. *Agricultural and Forest Meteorology*, 142(2-4):120-132.

Lassey, K.R., C.S. Pinares-Patiño, R.J. Martin, G. Molano, et al. 2011. Enteric methane emission rates determined by the SF6 tracer technique: Temporal patterns and averaging periods. *Animal Feed Science and Technology*, 166–167(0):183-191.

Laubach, J., and F.M. Kelliher. 2005. Measuring methane emission rates of a dairy cow herd (II): results from a backward-Lagrangian stochastic model. *Agricultural and Forest Meteorology*, 129(3-4):137-150.

Laubach, J., F.M. Kelliher, T.W. Knight, H. Clark, et al. 2008. Methane emissions from beef cattle—a comparison of paddock- and animal-scale measurements. *Australian Journal of Experimental Agriculture*, 48:132-137.

Le, P., A. Aarnink, A. Jongbloed, C. van der Peet Schwering, et al. 2008. Interactive effects of dietary crude protein and fermentable carbohydrate levels on odour from pig manure. *Livestock Science*, 114:48–61.

Le, P.D., A.J.A. Aarnink, A.W. Jongbloed, C.M.C. van der Peet Schwering, et al. 2007. Effects of crystalline amino acid supplementation to the diet on odor from pig manure. *Journal of Animal Science*, 85(3):791-801.

Lee, S.S., J.-T. Hsu, H.C. Mantovani, and J.B. Russell. 2002. The effect of bovicin HC5, a bacteriocin from Streptococcus bovis HC5, on ruminal methane production in vitro1. *FEMS Microbiology Letters*, 217(1):51-55.

Legesse, G., J.A. Small, S.L. Scott, G.H. Crow, et al. 2011. Predictions of enteric methane emissions for various summer pasture and winter feeding strategies for cow calf production. *Animal Feed Science and Technology*, 166-167(0):678-687.

Lenis, N. 1993. Lower nitrogen excretion in pig husbandry by feeding: Current and future possibilities. Proceedings of the First Inter. Symp. Nitrogen Flow in Pig Production and Environmental Consequences., Pudoc, Wageningen, Netherlands.

Leytem, A.B., R.S. Dungan, D.L. Bjorneberg, and A.C. Koehn. 2011. Emissions of Ammonia, Methane, Carbon Dioxide, and Nitrous Oxide from Dairy Cattle Housing and Manure Management Systems. *Journal of Environmental Quality*, 40(5):1383-1394.

Leytem, A.B., R.S. Dungan, D.L. Bjorneberg, and A.C. Koehn. 2013. Greenhouse gas and ammonia emissions from an open-freestall dairy in southern Idaho. *Journal of Environmental Quality*, 42:10-20.

Li, L., J. Cyriac, K.F. Knowlton, L.C. Marr, et al. 2009. Effects of reducing dietary nitrogen on ammonia emissions from manure on the floor of a naturally ventilated free stall dairy barn at low (0-20C) temperatures. *Journal of Environmental Quality*, 38:2172-2181.

Li, W., W.J. Powers, D. Karcher, C.R. Angel, et al. 2010. Effect of DDGS and mineral sources on air emissions from laying hens. *Poultry Science*, 89(E-Suppl. 1).

Li, W., W. Powers, and G.M. Hill. 2011. Feeding DDGS to swine and resulting impact on air emissions. *Journal of Animal Science*, 89:3286-3299.

Liebig, M.A., J.R. Gross, S.L. Kronberg, R.L. Phillips, et al. 2010. Grazing management contributions to net global warming potential: A long-term evaluation in the Northern Great Plains. *Journal of Environmental Quality*, 39:799-809.

Little, S., J. Linderman, K. MacLean, and H. Janzen. 2008. *Holos – a tool to estimate an dreduce greenhouse gases from farms. Methodology and algorithms for versions 1.1x*: Agriculture and Agri-Food Canada. http://www4.agr.gc.ca/AAFC-AAC/display-afficher.do?id=1226606460726&lang=eng#s1.

Liu, Z., H. Liu, and W. Powers. 2011a. Meta-analysis of Greenhouse Gas Emissions from swine operations. Proceedings of the ASABE Annual Meeting, August 7-10, 2011, Louisville, KY.

Liu, Z., W. Powers, D. Karcher, R. Angel, et al. 2011b. Effect of amino acid formulation and supplementation on nutrient mass balance in turkeys. *Poultry Science*, 90(6):1153-1161.

Liu, Z., W. Powers, D. Karcher, R. Angel, et al. 2011a. Effect of amino acid formulation and supplementation on air emissions from turkeys. *Trans. ASABE*, 54:617-628.

Locke, M.A., M.A. Weaver, R.M. Zablotowicz, R.W. Steinriede, et al. 2011. Constructed wetlands as a component of the agricultural landscape: Mitigation of herbicides in simulated runoff from upland drainage areas. *Chemosphere*.

Loh, Z., D. Chen, M. Bai, T. Naylor, et al. 2008. Measurement of greenhouse gas emissions from Australian feedlot beef production using open-path spectroscopy and atmospheric dispersion modeling. *Australian Journal of Experimental Agriculture*, 48:244-247.

Lovanh, N., J. Warren, and K. Sistani. 2010. Determination of ammonia and greenhouse gas emissions from land application of swine slurry: A comparison of three application methods. *Bioresource Technology*, 101(6):1662-1667.

Lovett, D., S. Lovell, L. Stack, J. Callan, et al. 2003. Effect of forage/concentrate ratio and dietary coconut oil level on methane output and performance of finishing beef heifers. *Livestock Production Science*, 84(2):135-146.

Lu, S.Y., P.Y. Zhang, and W.H. Cui. 2010. Impact of plant harvesting on nitrogen and phosphorus removal in constructed wetlands treating agricultural region wastewater. *International Journal of Environment and Pollution*, 43(4):339-353.

Luo, J., and S. Saggar. 2008. Nitrous oxide and methane emissions from a dairy farm stand-off pad. *Australian Journal of Experimental Agriculture*, 48:179-182.

Lupo, C.D., D.E. Clay, J.L. Benning, and J.J. Stone. 2013. Life-Cycle Assessment of the Beef Cattle Production System for the Northern Great Plains, USA. *Journal of Environmental Quality*, 42(5):1386-1394.

Malone, R.W., L. Ma, P. Heilman, D.L. Karlen, et al. 2007. Simulated N management effects on corn yield and tile-drainage nitrate loss. *Geoderma*, 140(3):272-283.

Maltais-Landry, G., R. Maranger, J. Brisson, and F. Chazarenc. 2009. Greenhouse gas production and efficiency of planted and artificially aerated constructed wetlands. *Environmental Pollution*, 157(3):748-754.

Mander, U., K. Lõhmus, S. Teiter, K. Nurk, et al. 2005a. Gaseous fluxes from subsurface flow constructed wetlands for wastewater treatment. *Journal of Environmental Science and Health - Part A Toxic/Hazardous Substances and Environmental Engineering*, 40(6-7):1215-1226.

Mander, U., S. Teiter, and J. Augustin. 2005b. Emission of greenhouse gases from constructed wetlands for wastewater treatment and from riparian buffer zones.

Martin, C., D.P. Morgavi, and M. Doreau. 2010. Methane mitigation in ruminants: from microbe to the farm scale. *Animal*, 4(03):351-365.

McGinn, S.M., H.H. Janzen, and T. Coates. 2003. Atmospheric Ammonia, Volatile Fatty Acids, and Other Odorants near Beef Feedlots. *Journal of Environmental Quality*, 32(4):1173-1182.

McGinn, S.M., K.A. Beauchemin, T. Coates, and D. Colombatto. 2004. Methane emissions from beef cattle: Effects of monensin, sunflower oil, enzymes, yeast, and fumaric acid. *Journal of Animal Science*, 82(11):3346-3356.

McGinn, S.M., K.A. Beauchemin, A.D. Iwaasa, and T. McAllister. 2006. Assessment of th sulfur hexafluoride (SF6) tracer technique for measuring enteric methane emissions from catttle. *Journal of Environmental Quality*, 35:1686-1691.

McGinn, S.M., D. Chen, Z. Loh, J. Hill, et al. 2008. Methane emissions from feedlot cattle in Australia and Canada. *Australian Journal of Experimental Agriculture*, 48:183-185.

McGinn, S.M., Y.-H. Chung, K.A. Beauchemin, A.D. Iwaasa, et al. 2009. Use of corn distillers' dried grains to reduce enteric methane loss from beef cattle. *Canadian Journal of Animal Science*, 89(3):409-413.

McGinn, S.M., D. Turner, N. Tomkins, E. Charmley, et al. 2011. Methane Emissions from Grazing Cattle Using Point-Source Dispersion. *Journal of Environmental Quality*, 40(1):22-27.

Michal, J.J., E. Allwine, S. Spogen, S. Pressley, et al. 2010. Nitrous oxide and methane emissions form a large beef feedlot. Proceedings of the Greenhouse Gas in Animal Agriculture Conference, Banff, Alberta, Canada.

Milano, G., and H. Clark. 2008. The effect of level of intake and forage quality on methane production by sheep. *Australian Journal of Experimental Agriculture*, 48:219-222.

Miles, D., P. Owens, and D. Rowe. 2006. Spatial variability of litter gaseous flux within a commercial broiler house: ammonia, nitrous oxide, carbon dioxide, and methane. *Poultry Science*, 85(2):167-172.

Miles, D.M., D.E. Rowe, and P.R. Owens. 2008. Winter broiler litter gases and nitrogen compounds: Temporal and spatial trends. *Atmospheric Environment*, 42(14):3351-3363.

Miles, D.M., D.E. Rowe, and T.C. Cathcart. 2011. Litter ammonia generation: Moisture content and organic versus inorganic bedding materials. *Poultry Science*, 90(6):1162-1169.

Mills, J.A., J. Dijkstra, A. Bannink, S.B. Cammell, et al. 2001. A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: model development, evaluation, and application. *Journal of Animal Science*, 79(6):1584-1597.

Mills, J.A.N., E. Kebreab, C.M. Yates, L.A. Crompton, et al. 2003. Alternative approaches to predicting methane emissions from dairy cows. *Journal of Animal Science*, 81(12):3141-3150.

Miner, J.R. 1975. *Evaluation of alternative approaches to control odors from animal feedlots*. Moscow, ID: Idaho Research Foundation.

Misselbrook, T., D. Chadwick, B. Pain, and D. Headon. 1998. Dietary manipulation as a means of decreasing N losses and methane emissions and improving herbage N uptake following application of pig slurry to grassland. *The Journal of Agricultural Science*, 130:183-191.

Misselbrook, T.H., J.M. Powell, G.A. Broderick, and J.H. Grabber. 2005. Dietary Manipulation in Dairy Cattle: Laboratory Experiments to Assess the Influence on Ammonia Emissions. *Journal of Dairy Science*, 88(5):1765-1777.

Moe, P.W., and H.F. Tyrrell. 1979. Methane Production in Dairy Cows. *Journal of Dairy Science*, 62(10):1583-1586.

Møller, H.B., S.G. Sommer, and B.K. Ahring. 2004. Methane productivity of manure, straw and solid fractions of manure. *Biomass and Bioenergy*, 26(5):485-495.

Møller, H.G., I. Lund, and S.G. Sommer. 2000. Solid-liquid separation of livestock slurry: efficiency and cost. *Bioresource Technology*, 74(2000):223-229.

Monteny, G.-J., A. Bannink, and D. Chadwick. 2006. Greenhouse gas abatement strategies for animal husbandry. *Agriculture, Ecosystems & Environment*, 112(2-3):163-170.

Montgomery, J.L., C.R. Krehbiel, J.J. Cranston, D.A. Yates, et al. 2009. Dietary zilpaterol hydrochloride. I. Feedlot performance and carcass traits of steers and heifers. *Journal of Animal Science*, 87:1374-1383.

Moore, P.A., T.C. Daniel, and D.R. Edwards. 2000. Reducing phosphorus runoff and inhibiting ammonia loss from poultry manure with aluminum sulfate. *Journal of Environmental Quality*, 29(37-49).

Moore, P.A., D. Miles, R. Burns, D. Pote, et al. 2011. Ammonia emission factors from broiler litter in barns, in storage, and after land application. *Journal of Environmental Quality*, 40:1395-1404.

Moore, P.A. 2013. *Treating poultry litter with aluminum Sulfate (Alum)*: Emission Management Practices Fact Sheet. USDA Livestock GRACEnet, U.S. Department of Agriculture.

Moore, P.A.J., D. Miles, R. Burns, D. Pote, et al. 2010. Ammonia emission factors from boiler litter in barns, in storage, and after land application. *Journal of Environmental Quality*, 40:1395-1404.

Moore, S.S., F.D. Mujibi, and E.L. Sherman. 2009. Molecular basis for residual feed intake in beef cattle. *Journal of Animal Science*, 87(14 suppl):E41-E47.

Morgavi, D.P., E. Forano, C. Martin, and C.J. Newbold. 2010. Microbial ecosystem and methanogenesis in ruminants. *Animal*, 4(Special Issue 07):1024-1036.

Münger, A., and M. Kreuzer. 2008. Absence of persistent methane emission differences in three breeds of dairy cows. *Australian Journal of Experimental Agriculture*, 48:77-82.

Murphy, M.R., R.L. Baldwin, and L.J. Koong. 1982. Estimation of Stoichiometric Parameters for Rumen Fermentation of Roughage and Concentrate Diets. *Journal of Animal Science*, 55(2):411-421.

Mustafa, A., M. Scholz, R. Harrington, and P. Carroll. 2009. Long-term performance of a representative integrated constructed wetland treating farmyard runoff. *Ecological Engineering*, 35(5):779-790.

Ndegwa, P., A. Hristov, J. Arogo, and R. Sheffield. 2008. A Review of Ammonia Emission Techniques for Concentrated Animal feeding Operations. *Biosystems Engineering*, 100:453-469.

New Zealand Ministry for the Environment. 2010. *Projected balance of emissions units during the first commitment period of the Kyoto Protocol*. http://www.mfe.govt.nz/publications/climate/projected-balance-units-may05/html/page10.html.

NGGIC. 1996. *Australian Methodology for the Estimation of Greenhouse Gas Emissions and Sinks. Agriculture, Workbook for Livestock, Workbook 6.1, Revision 1*. Canberra, Australia: National Greenhouse Inventory Committee, Department of the Environment, Sport, and Territories.

Ngwabie, N.M., K.H. Jeppsson, S. Nimmermark, C. Swensson, et al. 2009. Multi-location measurements of greenhouse gases and emission rates of methane and ammonia from a naturally-ventilated barn for dairy cows. *Biosystems Engineering*, 103(1):68-77.

Ni, J.Q. 1999. Mechanistic models of ammonia release from liquid manure: a review. *Journal of Agricultural Engineering Research*, 72(1):1-17.

Nielsen, D.A., L.P. Nielsen, A. Schramm, and N.P. Revsbech. 2010. Oxygen Distribution and Potential Ammonia Oxidation in Floating, Liquid Manure Crusts. *Journal of Environmental Quality*, 39(5):1813-1820.

Nkrumah, J.D., E.K. Okine, G.W. Mathison, K. Schmid, et al. 2006. Relationships of feedlot feed efficiency, performance, and feeding behavior with metabolic rate, methane production, and energy partitioning in beef cattle. *Journal of Animal Science*, 84(1):145-153.

Nozière, P., I. Ortigues-Marty, C. Loncke, and D. Sauvant. 2010. Carbohydrate quantitative digestion and absorption in ruminants: from feed starch and fibre to nutrients available for tissues. *Animal*, 4(Special Issue 07):1057-1074.

NRC. 1989. *Nutrient Requirements of Dairy Cattle*. Washington, DC: National Research Council, National Academy of Science,.

NRC. 2000. *Nutrient Requirements of Beef Cattle Update 2000*. Washington, DC: Natl. Acad. Press.

Ocfemia, K.S., Y. Zhang, and T. Funk. 2006. Hydrothermal processing of swine manure to oil using a continuous reactor system: Effects of operating parameters on oil yield and quality. *Transactions of the ASABE*, 49(6):1897-1904.

Odongo, N.E., R. Bagg, G. Vessie, P. Dick, et al. 2007. Long-Term Effects of Feeding Monensin on Methane Production in Lactating Dairy Cows. *Journal of Dairy Science*, 90(4):1781-1788.

Olson, K.C., J.A. Walker, C.A. Stonecipher, B.R. Bowman, et al. 2000. Effect of grass species on methane emissions by beef cattle. Soc. Range Manage. Annual Meeting.

Ominski, K.H., D.A. Boadi, and K.M. Wittenberg. 2006. Enteric methane emissions from backgrounded cattle consuming all-forage diets. *Canadian Journal of Animal Science*, 86(3):393-400.

Outor-Monteiro, D., V.M. Carvalho Pinheiro, L.J. Medeiros Mourão, and M.A. Machado Rodrigues. 2010. Strategies for mitigation of nitrogen environmental impact from swine production. *R. Bras. Zootec*, 39:317-325.

Owens, F.N., D.S. Secrist, W.J. Hill, and D.R. Gill. 1997. The effect of grain source and grain processing on performance of feedlot cattle: A review. *75*(868-879).

Panetta, D.M., W.J. Powers, H. Xin, B. Kerr, et al. 2006. Nitrogen excretion and ammonia emissions from pigs fed modified diets. *Journal of Environmental Quality*, 35(4):1297-1308.

Parker, D.B., E.A. Caraway, M.B. Rhoades, N.A. Cole, et al. 2010. Effect of wind tunnel air velocity on VOC flux from standard solutions and CAFO manure/wastewater. *Trans. ASABE*, 53:831-845.

Paul, J.W., N.E. Dinn, T. Kannangara, and L.J. Fisher. 1998. Protein Content in Dairy Cattle Diets Affects Ammonia Losses and Fertilizer Nitrogen Value. *Journal of Environmental Quality*, 27(3):528-534.

Paul, R., and W.W. Watson. 1966. THERMAL DIFFUSION AND SELF-DIFFUSION IN AMMONIA. *Journal of Chemical Physics*, 45(7):2675-&.

Pavao-Zuckerman, M.A., J.C. Waller, T. Ingle, and H.A. Fribourg. 1999. Methane Emissions of Beef Cattle Grazing Tall Fescue Pastures at Three Levels of Endophyte Infestation. *Journal of Environmental Quality*, 28(6):1963-1969.

Pelletier, N., R. Pirog, and R. Rasmussen. 2010. Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. *Agricultural Systems*, 103:380-389.

Peters, G.M., H.V. Rowley, S. Wiedemann, R. Tucker, et al. 2010. Red meat production in Australia: Life cycle assessment and comparison with overseas studies. *Environmental Science and Technology*, 44:1327-1332.

Petersen, S.O., and S.G. Sommer. 2011. Ammonia and nitrous oxide interactions: Roles of manure organic matter management. *Animal Feed Science and Technology*, 166–167:503-513.

Phetteplace, H., D. Johnson, and A. Seidl. 2001. Greenhouse gas emissions from simulated beef and dairy livestock systems in the United States. *Nutrient Cycling in Agroecosystems*, 60(1):99-102.

Philippe, F.-X., M. Laitat, B. Canart, M. Vandenheede, et al. 2007. Comparison of ammonia and greenhouse gas emissions during the fattening of pigs, kept either on fully slatted floor or on deep litter. *Livestock Science*, 111:144-152.

Picek, T., H. Čížková, and J. Dušek. 2007. Greenhouse gas emissions from a constructed wetland-Plants as important sources of carbon. *Ecological Engineering*, 31(2):98-106.

Pinares-Patiño, C.S., M.J. Ulyatt, G.C. Waghorn, K.R. Lassey, et al. 2003. Methane emission by alpaca and sheep fed on lucerne hay or grazed on pastures of perennial ryegrass/white clover or birdsfoot trefoil. *Journal of Agricultural Science*, 140(02):215-226.

Portejoie, S., J. Dourmad, J. Martinez, and Y. Lebreton. 2004. Effect of lowering crude protein on nitrogen excretion, manure composition and ammonia emission from fattening pigs. . *Livestock Prod. Sci.*, 91:45-55.

Pos, J., R. Trapp, and M. Harvey. 1984. *Performance of a Brushed Screen / Roller Press Manure Separator*, ASAE Paper No. 83-4065. St. Joseph, MI: American Society of Agricultural Engineers.

Powell, J.M., P.R. Cusick, T.H. Misselbrook, and B.J. Holmes. 2007. Design and calibration of chambers for measuring ammonia emissions from tie-stall dairy barns. *Trans. ASABE*, 49(4):1139-1149.

Powell, J.M., G.A. Broderick, and T.H. Misselbrook. 2008. Seasonal Diet Affects Ammonia Emissions from Tie-Stall Dairy Barns. *Journal of Dairy Science*, 91(2):857-869.

Powell, J.M., G.A. Broderick, J.H. Grabber, and U.C. Hymes-Fecht. 2009. Technical note: Effects of forage protein-binding polyphenols on chemistry of dairy excreta. *Journal of Dairy Science*, 92(4):1765-1769.

Powell, J.M., M.J. Aguerre, and M.A. Wattiaux. 2011. Tannin Extracts Abate Ammonia Emissions from Simulated Dairy Barn Floors. *Journal of Environmental Quality*, 40(3):907-914.

Powers, W., S. Zamzow, and B. Kerr. 2007. Reduced crude protein effects on aerial emissions from swine. *Applied Engineering in Agriculture*, 23:539-546.

Powlson, D.S., A.B. Riche, K. Coleman, M.J. Glendining, et al. 2008. Carbon sequestration in European soils through straw incorporation: Limitations and alternatives. *Waste Management*, 28(4):741-746.

Preston, R.L. 2013. Nutrient Values for 300 Cattle Feeds. *Beef Magazine*.

Radunz, A. 2011. *Optaflexx and Zilmax: Beta agonists: Growth promoting feed additives for beef cattle*: University of Wyoming Extension Report.

Raman, K.P., W.P. Walawender, and L.T. Fan. 1980. Gasification of feedlot manure in a fluidized bed reactor. The effect of temperature. *Industrial & Engineering Chemistry Process Design and Development*, 19(4):623-629.

Reynolds, C.K., J.A.N. Mills, L.A. Crompton, D.I. Givens, et al. 2010. Ruminant nutrition regimes to reduce greenhouse gas emissions in dairy cows. In *Energy and protein metabolism and nutrition*, G. M. Crovetto (ed.): EEAP

Ro, K.S., K. Cantrell, D. Elliott, and P.G. Hunt. 2007. Catalytic wet gasification of municipal and animal wastes. *Industrial and Engineering Chemistry Research*, 46(26):8839-8845.

Ro, K.S., K.B. Cantrell, P.G. Hunt, T.F. Ducey, et al. 2009. Thermochemical conversion of livestock wastes: Carbonization of swine solids. *Bioresource Technology*, 100(22):5466-5471.

Ro, K.S., K.B. Cantrell, and P.G. Hunt. 2010. High-temperature pyrolysis of blended animal manures for producing renewable energy and value-added biochar. *Industrial and Engineering Chemistry Research*, 49(20):10125-10131.

Roberts, S.A., H. Xin, B.J. Kerr, J.R. Russell, et al. 2007. Effects of Dietary Fiber and Reduced Crude Protein on Ammonia Emission from Laying-Hen Manure. *Poultry Science*, 86(8):1625-1632.

Robinson, B., and E. Okine. 2001. *Feed intake in feedlot cattle. Alberta Feedlot Management Guide, 2nd Edition*. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/beef4873.

Rotz, C.A. 2004. Management to reduce nitrogen losses in animal production. *Journal of Animal Science*, 82(13 suppl):E119-E137.

Rotz, C.A., D.R. Buckmaster, and J.W. Comerford. 2005. A beef herd model for simulating feed intake, animal performance, and manure excretion in farm systems. *Journal of Animal Science*, 83(1):231-242.

Rotz, C.A., D.S. Chianese, F. Montes, and S. Hafner. 2011a. *Dairy gas emissions model: Reference manual*: U.S. Department of Agriculture, Agricultural Research Service.

Rotz, C.A., M.S. Corson, D.S. Chianese, F. Montes, et al. 2011b. *Integrated farm system model: Reference Manual*. University Park, PA: U.S. Department of Agriculture, Agricultural Research Service. http://ars.usda.gov/SP2UserFiles/Place/19020000/ifsmreference.pdf.

Saggar, S., C.B. Hedley, D.L. Giltrap, and S.M. Lambie. 2007. Measured and modelled estimates of nitrous oxide emission and methane consumption from a sheep-grazed pasture. *Agriculture, Ecosystems & Environment*, 122(3):357-365.

Saha, C.K., C. Ammon, W. Berg, M. Fiedler, et al. 2014. Seasonal and diel variations of ammonia and methane emissions from a naturally ventilated dairy building and the associated factors influencing emissions. *Science of the Total Environment*, 468:469-462.

Samer, M., M. Fiedler, H.J. Müller, M. Gläser, et al. 2011. Winter measurements of air exchange rates using tracer gas technique and quantification of gaseous emissions from a naturally ventilated dairy barn. *Applied Engineering in Agriculture*.

Seo, D.C., and R.D. DeLaune. 2010. Fungal and bacterial mediated denitrification in wetlands: Influence of sediment redox condition. *Water Research*, 44(8):2441-2450.

Shiflett, J.S. 2011. *Sheep and Lamb Industry Economic Impact Analysis*: American Sheep Industry Association http://www.sheepusa.org/user_files/file_865.pdf

Shutt, J.W., R.K. White, E.P. Taiganides, and C.R. Mote. 1975. Evaluation of Solids Separation Devices. Proceedings of the 3rd International Symposium on Livestock Wastes, Urbana- Champaign, IL, April 21-24, 1975.

Sneath, R.W., M. Shaw, and A.G. Williams. 1988. Centrifugation for separating piggery slurry 1. The performance of a decanting centrifuge. *Journal of Agricultural Engineering Research*, 39(3):181-190.

Sommer, S.G., S.O. Petersen, and H.B. Møller. 2004. Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutrient Cycling in Agroecosystems*, 69:143-154.

Soosaar, K., M. Maddison, and Ü. Mander. 2009. Water quality and emission rates of greenhouse gases in a treatment reedbed.

Søvik, A.K., J. Augustin, K. Heikkinen, J.T. Huttunend, et al. 2006. Emission of the Greenhouse Gases Nitrous Oxide and Methane from Constructed Wetlands in Europe. *Journal of Environmental Quality*, 35(6):2360-2373.

Spiehs, M.J., B.L. Woodbury, B.E. Doran, R.A. Eigenberg, et al. 2011. Environmental conditions in deep-bedded mono-slope facilities: A descriptive study. *Trans. ASABE*, 54:663-673.

Stackhouse-Lawson, K.R., C.A. Rotz, J.W. Oltjen, and F.M. Mitloehner. 2012. Carbon footprint and ammonia emissions of California beef production systems. *Journal of Animal Science*, 90:4641-4655.

Stackhouse, K.R., C.A. Rotz, J.W. Oltjen, and F.M. Mitloehner. 2012. Growth promoting technologies reduce the carbon footprint, ammonia emissions, and cost of California beef production systems. *Journal of Animal Science*, 90:4656-4665.

Stein, O.R., and P.B. Hook. 2005. Temperature, plants, and oxygen: How does season affect constructed wetland performance? *Journal of Environmental Science and Health - Part A Toxic/Hazardous Substances and Environmental Engineering*, 40(6-7):1331-1342.

Stein, O.R., J.A. Biederman, P.B. Hook, and W.C. Allen. 2006. Plant species and temperature effects on the k-C* first-order model for COD removal in batch-loaded SSF wetlands. *Ecological Engineering*, 26(2):100-112.

Stein, O.R., B.W. Towler, P.B. Hook, and J.A. Biederman. 2007a. On fitting the k-C* first order model to batch loaded sub-surface treatment wetlands. *Water Science and Technology*, 56(3):93-99.

Stein, O.R., B.W. Towler, P.B. Hook, and J.A. Biederman. 2007b. On fitting the k-C* first order model to batch loaded sub-surface treatment wetlands.

Stone, K.C., P.G. Hunt, A.A. Szögi, F.J. Humenik, et al. 2002. Constructed wetland design and performance for swine lagoon wastewater treatment. *Transactions of the American Society of Agricultural Engineers*, 45(3):723-730.

Stone, K.C., M.E. Poach, P.G. Hunt, and G.B. Reddy. 2004. Marsh-pond-marsh constructed wetland design analysis for swine lagoon wastewater treatment. *Ecological Engineering*, 23(2):127-133.

Stone, K.C., P.G. Hunt, K.B. Cantrell, and K.S. Ro. 2010. The potential impacts of biomass feedstock production on water resource availability. *Bioresource Technology*, 101(6):2014-2025.

Sun, H., S.L. Trabue, K. Scoggin, W.A. Jackson, et al. 2008. Alcohol, Volatile Fatty Acid, Phenol, and Methane Emissions from Dairy Cows and Fresh Manure. *Journal of Environmental Quality*, 37(2):615-622.

Sutton, A., K. Kephardt, J. Patterson, R. Mumma, et al. 1996. Manipulating swine diets to reduce ammonia and odor emissions. Proceedings of the 1st International Conference on Air Pollution from Agricultural Operations, Kansas City, MO.

Sveinbjornsson, P., P. Huhtanen, and J. Uden. 2006. The Nordic dairy cow model, Karoline – development of volatile fatty acid submodel. In *Nutrient Digestion and Utilization in Farm Animals: Modeling Approach*, E. Kebreab, J. Dijkstra, A. Bannink, W. J. J. Gerrits and J. France (eds.). Wallingford, UK: CAB Publishing.

Sweeten, J. 2004. *Air Quality: Odor, Dust, and Gaseous Emissions from Concentrated Animal Feeding Operations in the Southern Great Plains*, Project No. 2003-34466-13146 / CSREES Project # TS-2003-06007: U.S. Department of Agriculture, CSREES Special Research Grants Program

Szanto, G.L., H.V.M. Hamelers, W.H. Rulkens, and A.H.M. BVeeken. 2006. NH_3, N_2O and CH_4 emissions during passively aerated composting of straw-rich pig manure. *Bioresource Technology*, 98:2659-2670.

Tanner, C.C., D.D. Adams, and M.T. Downes. 1997. Methane emissions from constructed wetlands treating agricultural wastewaters. *Journal of Environmental Quality*, 26(4):1056-1062.

Tanner, C.C., and T.R. Headley. 2011. Components of floating emergent macrophyte treatment wetlands influencing removal of stormwater pollutants. *Ecological Engineering*, 37(3):474-486.

Tanner, C.C., and J.P.S. Sukias. 2011. Multiyear nutrient removal performance of three constructed wetlands intercepting tile drain flows from grazed pastures. *Journal of Environmental Quality*, 40(2):620-633.

Taylor, C.R., P.B. Hook, O.R. Stein, and C.A. Zabinski. 2010. Seasonal effects of 19 plant species on COD removal in subsurface treatment wetland microcosms. *Ecological Engineering*.

Tedeschi, L.O., D.G. Fox, and T.P. Tylutki. 2003. Potential Environmental Benefits of Ionophores in Ruminant Diets. *Journal of Environmental Quality*, 32(5):1591-1602.

Teiter, S., and U. Mander. 2005. Emission of N2O, N2, CH4, and CO 2 from constructed wetlands for wastewater treatment and from riparian buffer zones. *Ecological Engineering*, 25(5):528-541.

Todd, R.W., N.A. Cole, L.A. Harper, T.K. Flesch, et al. 2005. Ammonia and gaseous nitrogen emissions from a commercial beef cattle feedyard estimated using the flux-gradient method and N/P ratio analysis. Proceedings of the State of the Science: Animal manure and waste management, Jan 5-7, 2005, National Center for manure and Waste Management, San Antonio, TX.

Todd, R.W., N.A. Cole, H.M. Waldrip, and R.M. Aiken. 2013. Arrhenius equation for modeling feedyard ammonia emission using temperature and diet crude protein. *Journal of Environmental Quality*, 42:666-671.

Todd, R.W., M. Altman, N.A. Cole, and H.M. Waldrip. 2014a. Methane emissiosns from a beef cattle feedyard during winter and summer on the Southern High Plains of Texas. *Journal of Environmental Quality*(in press).

Todd, R.W., H.M. Waldrip, M. Altman, and N.A. Cole. 2014b. Methane emissions from a beef cattle feedyard: measurements and models. Proceedings of the American Meteorological Society's 31st Conference on Agricultural and Forest Meteolorology, May 12-15, 2014, Portland, OR.

Tomkins, N.W., and R.A. Hunter. 2004. Methane mitigation in beef cattle using a patented anti-methanogen. Proceedings of the 2nd Joint Australia and New Zealand Forum on Non-CO2 Greenhouse Gas Emissions from Agriculture, October 2003, Lancemore Hill, Canberra.

Tomkins, N.W., S.M. Colegate, and R.A. Hunter. 2009. A bromochloromethane formulation reduces enteric methanogenesis in cattle fed grain-based diets. *Animal Production Science*, 49(12):1053-1058.

Tomkins, N.W., S.M. McGinn, D.A. Turner, and E. Charmley. 2011. Comparison of open-circuit respiration chambers with a micrometeorological method for determining methane emissions from beef cattle grazing a tropical pasture. *Animal Feed Science and Technology*, 166-167(0):240-247.

Towler, B.W., J.E. Cahoon, and O.R. Stein. 2004. Evapotranspiration crop coefficients for cattail and bulrush. *Journal of Hydrologic Engineering*, 9(3):235-239.

Trei, J.E., G.C. Scott, and R.C. Parish. 1972. Influence of Methane Inhibition on Energetic Efficiency of Lambs. *Journal of Animal Science*, 34(3):510-515.

U.S. EPA. 2011. *U.S. GHG Inventory 1990-2009*: U.S. Envrionmental Protection Agency. http://www.epa.gov/climatechange/emissions/downloads11/US-GHG-Inventory-2011-Chapter-6-Agriculture.pdf.

U.S. EPA. 2013. *Inventory of U.S. greenhouse gas emissions and sinks: 1990-2011*. Washington, D.C.: Environmental Protection Agency.

Ungerfeld, E.M., R.A. Kohn, R.J. Wallace, and C.J. Newbold. 2007. A meta-analysis of fumarate effects on methane production in ruminal batch cultures. *Journal of Animal Science*, 85(10):2556-2563.

USDA. 2004a. *Dairy 2002. Nutrient Management and the U.S. Dairy Industry in 2002*. Washington, DC: USDA Animal and Plant Health Inspection Service. http://nahms.aphis.usda.gov/dairy/dairy02/Dairy02Nutrient_mgmt_rept.pdf.

USDA. 2004b. *USDA Agriculture and Forestry Greenhouse Gas Inventory: 1990-2001*. Washington, DC: U.S. Department of Agriculture. http://www.usda.gov/oce/climate_change/ghg_inventory.htm.

USDA. 2010. *Beef 2007-08, Part V: Reference of Beef Cow-calf management practices in the United States, 2007-2008*. Fort Collins, CO: USDA-APHIS-VS, CEAH.

USDA NASS. 2011. *Charts and Maps- Sheep and Lamb*: U.S. Department of Agriculture, National Agricultural Statistics Service. http://www.nass.usda.gov/Charts_and_Maps/Sheep_and_Lambs/index.asp.

USDA NASS. 2012. *Quick Stats: Agricultural Statistics Database*. Washington, DC: U.S. Department of Agriculture, National Agriculture Statistics Service. http://quickstats.nass.usda.gov/.

USDA NRCS. 2007. *Composting Manure – What's going on in the dark?*, May 2007, Number 1. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service.

Van Kessel, J.A.S., and J.B. Russell. 1996. The effect of pH on ruminal methanogenesis. *FEMS Microbiology Ecology*, 20(4):205-210.

VanAmburgh, M., L. Chase, T. Overton, E. Recktenwald, et al. 2010. 2010 Updates to the Cornell Net Carbohydrate and Protein System v6.1. Proceedings of the Cornell Nutrition Conference.

Vander Pol, K.J., M.K. Luebbe, G.I. Crawford, G.E. Erickson, et al. 2009. Performance and digestibility characteristics of finishing diets containing distillers grains, composites of corn processing coproducts, or supplemental corn oil. *Journal of Animal Science*, 87(2):639-652.

VanderZaag, A.C., R.J. Gordon, D.L. Burton, R.C. Jamieson, et al. 2010. Greenhouse gas emissions from surface flow and subsurface flow constructed wetlands treating dairy wastewater. *Journal of Environmental Quality*, 39(2):460-471.

Vanotti, M.B., and P.G. Hunt. 1999. Solids and nutrient removal from flushed swine manure using polyacrylamides. *Transactions of the American Society of Agricultural Engineers*, 42(6):1833-1840.

Vanotti, M.B., and P.G. Hunt. 2000. Nitrification treatment of swine wastewater with acclimated nitrifying sludge immobilized in polymer pellets. *Transactions of the ASAE*, 43(2):405-413.

Vanotti, M.B., D.M.C. Rashash, and P.G. Hunt. 2002. Solid-liquid separation of flushed swine manure with PAM: Effect of wastewater strength. *Transactions of the American Society of Agricultural Engineers*, 45(6):1959-1969.

Vanotti, M.B., A.A. Szogi, and P.G. Hunt. 2003. Extraction of soluble phosphorus from swine wastewater. *Transactions of the American Society of Agricultural Engineers*, 46(6):1665-1674.

Vanotti, M.B., P.D. Millner, P.G. Hunt, and A.Q. Ellison. 2005. Removal of pathogen and indicator microorganisms from liquid swine manure in multi-step biological and chemical treatment. *Bioresource Technology*, 96(2):209-214.

Vanotti, M.B., A.A. Szogi, P.G. Hunt, P.D. Millner, et al. 2007. Development of environmentally superior treatment system to replace anaerobic swine lagoons in the USA. *Bioresource Technology*, 98(17):3184-3194.

Vanotti, M.B., and A.A. Szogi. 2008. Water quality improvements of wastewater from confined animal feeding operations after advanced treatment. *Journal of Environmental Quality*, 37(SUPPL. 5):S86-S96.

Vanotti, M.B., A.A. Szogi, and C.A. Vives. 2008. Greenhouse gas emission reduction and environmental quality improvement from implementation of aerobic waste treatment systems in swine farms. *Waste Management*, 28(4):759-766.

Vanotti, M.B., A.A. Szogi, P.D. Millner, and J.H. Loughrin. 2009. Development of a second-generation environmentally superior technology for treatment of swine manure in the USA. *Bioresource Technology*, 100(22):5406-5416.

Vanotti, M.B., Millner, P.D., Szogi, A.A., Campbell, C.R., Fetterman, L.M. . 2006. Aerobic composting of swine manure solids mixed with cotton gin waste. ASABE Annual International Meeting, Portland, Oregon.

Vasconcelos, J.T., and M.L. Galyean. 2007. Nutritional recommendations of feedlot consulting nutritionists: The 2007 Texas Tech University survey. *Journal of Animal Science*, 85(10):2772-2781.

Vasconcelos, J.T., R.J. Rathmann, R.R. Reuter, J. Leibovish, et al. 2008. Effects of duration of zilpaterol hydrochloride feeding and days of the finishing diet on feedlot cattle performance and carcass traits. *Journal of Animal Science*, 86:2005-2012.

Venterea, R.T., K.A. Spokas, and J.M. Baker. 2009. Accuracy and Precision Analysis of Chamber-Based Nitrous Oxide Gas Flux Estimates. *Soil Sci. Soc. Am. J.*, 73(4):1087-1093.

Venterea, R.T. 2010. Simplified Method for Quantifying Theoretical Underestimation of Chamber-Based Trace Gas Fluxes. *Journal of Environmental Quality*, 39(1):126-135.

Verge, X.P.C., J.A. Dyer, R.L. Desjardins, and D. Worth. 2008. Greenhouse gas emissions from the Canadian beef industry. *Agricultural Systems*, 98:126-134.

Verge, X.P.X., J.A. Dyer, R.L. Desjardins, and D. Worth. 2009. Greenhouse gas emissions from the Canadian pork industry. *Livestock Science*, 121:92-101.

Vogel, G. 1995. Effects of ionophores on feed intake by feedlot cattle. Proceedings of the Symposium on Intake of Feedlot Cattle, Oklahoma State Univ. July, 1995.

Vymazal, J. 2011. Enhancing ecosystem services on the landscape with created, constructed and restored wetlands. *Ecological Engineering*, 37(1):1-5.

Waghorn, G.C., H. Clark, V. Taufa, and A. Cavanagh. 2008. Monensin controlled-release capsules for methane mitigation in pasture-fed dairy cows. *Australian Journal of Experimental Agriculture*, 48:65-68.

Wagner, J.J., T.E. Engle, and T.C. Bryant. 2010. The effect of rumen degradable and rumen undegradable intake protein on feedlot performance and carcass merit in heavy yearling steers. *Journal of Animal Science*, 88:1073-1081.

Wang, L., A. Shahbazi, and M.A. Hanna. 2011. Characterization of corn stover, distiller grains and cattle manure for thermochemical conversion. *Biomass and Bioenergy*, 35(1):171-178.

Wang, Y., R. Inamori, H. Kong, K. Xu, et al. 2008. Influence of plant species and wastewater strength on constructed wetland methane emissions and associated microbial populations. *Ecological Engineering*, 32(1):22-29.

Westberg, H., B. Lamb, K.A. Johnson, and M. Huyler. 2001. Inventory of methane emissions from U. S. cattle. *Journal of Geophysical Research*, 106(D12):12633-12642.

White, F. 1999. *Fluid Mechanics*. Boston, MA: McGraw-Hill Science/Engineering/Math.

Wileman, B.W., D.U. Thomson, C.D. Reinhardt, and D.G. Renter. 2009. Analysis of modern technologies commonly used in beef cattle production: Conventional beef production versus nonconventional production using meta-analysis. *Journal of Animal Science*, 87:3418-3426.

Wilkerson, V.A., D.P. Casper, and D.R. Mertens. 1995. The Prediction of Methane Production of Holstein Cows by Several Equations. *Journal of Dairy Science*, 78(11):2402-2414.

Wolin, M.J. 1960. A Theoretical Rumen Fermentation Balance. *Journal of Dairy Science*, 43(10):1452-1459.

Woodbury, B.L., D. N. Miller, J. A. Nienaber, and R.A. Eigenberg. 2001. Seasonal and spatial variations of denitrifying enzyme activity in feedlot soil. *Trans. ASABE*, 44:1635-1642.

WRI. 2009. *Documentation of Emissions Calculations for Version 1.2 of the Manure and Nutrient Reduction Estimator (MANURE) Tool*. Arlington, VA: ERT - Winrock International. http://app6.erg.com/manure/docs/manure_calculations.pdf.

Wright, A.D.G., P. Kennedy, C.J. O'Neill, A.F. Toovey, et al. 2004. Reducing methane emissions in sheep by immunization against rumen methanogens. *Vaccine*, 22(29-30):3976-3985.

Wu-Haan, W., W.J. Powers, C.R. Angel, C.E. Hale, III, et al. 2007a. Nutrient Digestibility and Mass Balance in Laying Hens Fed a Commercial or Acidifying Diet. *Poultry Science*, 86(4):684-690.

Wu-Haan, W., W.J. Powers, C.R. Angel, C.E. Hale, III, et al. 2007b. Effect of an Acidifying Diet Combined with Zeolite and Slight Protein Reduction on Air Emissions from Laying Hens of Different Ages. *Poultry Science*, 86(1):182-190.

Wu, J., J. Zhang, W.L. Jia, H.J. Xie, et al. 2009. Nitrous oxide fluxes of constructed wetlands to treat sewage wastewater. *Huanjing Kexue/Environmental Science*, 30(11):3146-3151.

Xiu, S., Y. Zhang, and A. Shahbazi. 2009. Swine manure solids separation and thermochemical conversion to heavy oil. *BioResources*, 4(2):458-470.

Xiu, S., A. Shahbazi, C.W. Wallace, L. Wang, et al. 2011. Enhanced bio-oil production from swine manure co-liquefaction with crude glycerol. *Energy Conversion and Management*, 52(2):1004-1009.

Yan, T., R.E. Agnew, F.J. Gordon, and M.G. Porter. 2000. Prediction of methane energy output in dairy and beef cattle offered grass silage-based diets. *Livestock Production Science*, 64(2-3):253-263.

Yan, T., M.G. Porter, and C.S. Mayne. 2009. Prediction of methane emission from beef cattle using data measured in indirect open-circuit respiration calorimeters. *Animal*, 3(10):1455-1462.

Young, B.A. 1981. Cold stress as it affects animal production. *Journal of Animal Science*, 52:154-163.

Zhang, G., J.S. Strøm, B. Li, H.B. Rom, et al. 2005. Emission of ammonia and other contaminant gases from naturally ventilated dairy cattle buildings. *Biosystems Engineering*, 92(3):355-364.

Zhu, G., Z. Ma, Z. Gao, W. Ma, et al. 2014. Charaterizing CH4 and N2O emissions from an intensive dairy operation in summer and fall in China. *Atmospheric Environment*, 83:245-253.

Zhu, N., P. An, B. Krishnakumar, L. Zhao, et al. 2007. Effect of plant harvest on methane emission from two constructed wetlands designed for the treatment of wastewater. *Journal of Environmental Management*, 85(4):936-943.

Zinn, R.A., and Y. Shen. 1996. Interaction of dietary calcium and supplemental fat on digestive function and growth performance in feedlot steers. *Journal of Animal Science*, 74:2303-2309.

Zinn, R.A., and R. Barajas. 1997. Influence of flake density on the comparative feeding value of a barley-corn blend for feedlot cattle. *Journal of Animal Science*, 75(4):904-909.

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