TECHNICAL WORKING GROUP ON AGRICULTURAL GREENHOUSE GASES (T-AGG) REPORT

Near-Term Options for Reducing Greenhouse Gas Emissions from Livestock Systems in the United States

Beef, Dairy, and Swine Production Systems

Authors (in alphabetical order)

Shawn Archibeque[*]

Karen Haugen-Kozyra[†]

Kristen Johnson[‡]

Ermias Kebreab[§]

Wendy Powers-Schilling[**]

Edited by

Lydia P. Olander[††]

Abigail Van de Bogert[††]

[*] Colorado State University

[†] The Prasino Group, Edmonton, Alberta

[‡] Washington State University

[§] University of California, Davis

[**] Michigan State University

[††] Nicholas Institute for Environmental Policy Solutions, Duke University

NICHOLAS INSTITUTE FOR ENVIRONMENTAL POLICY SOLUTIONS

DUKE UNIVERSITY

July 2012

Nicholas Institute for Environmental Policy Solutions
Report
NI R 12-04
July 2012

Near-Term Options for Reducing Greenhouse Gas Emissions from Livestock Systems in the United States

Beef, Dairy, and Swine Production Systems

Authors (in alphabetical order)
Shawn Archibeque[*]
Karen Haugen-Kozyra[†]
Kristen Johnson[‡]
Ermias Kebreab[§]
Wendy Powers-Schilling[**]

Edited by
Lydia P. Olander[††]
Abigail Van de Bogert[††]

[*] *Colorado State University*
[†] *The Prasino Group, Edmonton, Alberta*
[‡] *Washington State University*
[§] *University of California, Davis*
[**] *Michigan State University*
[††] *Nicholas Institute for Environmental Policy Solutions, Duke University*

The editors gratefully acknowledge the David and Lucile Packard Foundation for its generous support of this work. The editors would also like to thank Nick Martin, Juan Tricarico, Garth Boyd, Gordon Carstens, and Xavier Vergé for their helpful reviews and feedback, Melissa Edeburn for her editing support, and Paul Brantley for his design skills and editing support.

NICHOLAS INSTITUTE
FOR ENVIRONMENTAL POLICY SOLUTIONS
DUKE UNIVERSITY

CONTENTS

LIST OF TABLES

LIST OF FIGURES

What is T-AGG?

GHG mitigation activities that increase carbon storage in soil or reduce methane and nitrous oxide emissions from agriculture can be an important part of U.S. and global climate change strategies. In November 2009, the Technical Working Group on Agricultural Greenhouse Gases (T-AGG) began assembling the scientific and analytical foundation to support implementation of high-quality agricultural greenhouse gas (GHG) reductions. At that time, only a few high-quality and widely approved methodologies for quantifying agricultural GHG benefits had been developed for various mitigation programs and markets. Many agricultural protocols are now published, and more are in development.

T-AGG is coordinated by the Nicholas Institute for Environmental Policy Solutions at Duke University. It works with academic and government scientists to build a foundational understanding of agricultural GHG accounting with the critical guidance of a broad range of experts and stakeholders. Its work is made possible by a grant from the David and Lucile Packard Foundation.

T-AGG has produced a series of reports that survey and prioritize agricultural mitigation opportunities in the United States and abroad with the goal of providing a roadmap for protocol and program development. The reports provide information necessary for designing agricultural GHG mitigation and reporting programs. They will be of use to private or voluntary markets and registries and commodity group and supply chain initiatives as well as regulatory agencies.

Coordinating Team

Project Director – Lydia Olander, *Director of Ecosystem Services Program, Nicholas Institute for Environmental Policy Solutions, and Research Scientist, Duke University*

Research Associate – Abigail Van De Bogert, *Research Analyst, Nicholas Institute, Duke University*

Advisors for the livestock project include Erin Fitzgerald of the Innovation Center for U.S. Dairy and the center's LCA expert Ying Wang; Marty Matlock, technical lead for The Sustainability Consortium, from the Center for Agricultural and Rural Sustainability at the University of Arkansas; and Garth Boyd, senior partner with The Prasino Group and the former senior vice president for Agriculture at Camco.

For more information visit http://www.nicholas.duke.edu/institute/t-agg.

ACRONYMS AND ABBREVIATIONS

ACR: American Carbon Registry
ALCA: attributional life-cycle assessment
ALU: Agricultural and Land Use National Greenhouse Gas Inventory
ANSI: American National Standards Institute
B_0: maximum methane producing capacity for manure
BAU: business-as-usual scenario (used to help assess impacts of GHG emissions)
C: carbon
C_2H_4: ethane
CAR: Climate Action Reserve
CAST: Council for Agricultural Science and Technology
CDM: Clean Development Mechanism
CEFM: Cattle Enteric Fermentation Model
CH_4: methane
CNCPS: Cornell Net Carbohydrate and Protein System
CO_2: carbon dioxide
CO_2e: carbon dioxide equivalent
CP: crude protein
DDGS: distillers' dried grains with solubles
DM: dry matter
EF: emission factors
Gg: gigagram
GHG: greenhouse gas
H_2: hydrogen gas
H_2S: hydrogen sulfide
IFSM: Integrated Farm Systems Model
IPCC: Intergovernmental Panel on Climate Change
ISO: International Organization for Standardization
LCI phase: life-cycle inventory analysis phase
LCIA phase: life-cycle impact assessment phase
MCF: methane conversion factors
N_2O: nitrous oxide
NAHMS: National Animal Health Monitoring System
NH_3: ammonia
OECD: Organisation for Economic Co-operation and Development
REC: renewable energy credit
RFI: residual feed intake
RIRDC: Rural Industries Research and Development Corporation (Australia)
SF_6: sulfur hexafluoride
Soil C: carbon stored in soil
t: tonne, or metric ton
Tg: Teragram
TSD: technical deed document
UNFAO or FAO: United Nations Food and Agriculture Organization
UNFCCC: United Nations Framework Convention on Climate Change
U.S. EPA: United States Environmental Protection Agency
USDA: United States Department of Agriculture
VCS: Verified Carbon Standard
VFA: volatile fatty acids
VS: volatile solid
WRI GHG Protocol: New greenhouse gas protocol developed by the World Resources Institute and the World Business Council for Sustainable Development
Ym: CH_4 conversion rates, expressed as the fraction of gross energy converted to CH_4

EXECUTIVE SUMMARY

The objective of this report, which is part of a series on agricultural greenhouse gas (GHG) emissions, is to synthesize and communicate the fundamental information necessary for designing agricultural GHG mitigation and reporting programs. The report will be of use to private or voluntary markets and registries, commodity group and supply chain initiatives, and regulatory agencies. It summarizes strategies for managing GHG emissions from livestock (cattle and swine) systems and reviews options for quantifying and accounting for farm-scale implementation of such strategies. It reviews the state of current knowledge in four areas:

1. Managing for GHG mitigation in dairy and beef production
2. Managing for GHG mitigation in swine production
3. Measurement and prediction of GHG emissions from livestock
4. Review of GHG accounting approaches for livestock systems

Sections 1 and 2 review what is known about the potential and limitations of a variety of mitigation strategies, including changing manure management, changing bedding systems, composting, using anaerobic digesters, and altering livestock diet. Any transition to widespread adoption of mitigation approaches will require a suite of incentives that lower costs and other barriers to adoption. Mitigation approaches frequently entail tradeoffs and the possibilities for leakage must be carefully considered. For both cattle and swine systems, anaerobic digesters could substantially mitigate emissions, but they come with significant infrastructure and maintenance costs.

Section 3 reviews methods for quantifying GHG emissions from livestock and assesses the strengths and weaknesses of different emissions models. Each model is designed for a specific application and each has its own limitations. For all of them, data are a limiting factor for wider application.

Section 4 reviews the use of life-cycle assessments (LCAs) and project-based accounting in quantification of agricultural GHG emissions. Intergovernmental Panel on Climate Change (IPCC) standards and methods are critical to these methods. It is important to note that LCAs and project-based accounting are not easily comparable and can vary significantly in analytical approach.

Projections of livestock and dairy production predict increases in productivity but not an increase in the number or extent of farms. Thus the focus in the United States will be on changing existing farm systems, not on creating new ones. This focus affects the costs and likelihood of mitigation strategies that work in existing systems rather than depend on new infrastructure.

A wide range of near-term livestock- and manure-management options can reduce emissions per unit of dairy, beef, and swine produced, but the cost-effectiveness of tying specific on-farm reductions to particular management practices remains uncertain. Farmers will require incentives if they are to justify management shifts. They will need to see the benefits of infrastructure or management changes in the form of revenues from primary products like energy production, from Farm Bill incentive programs, or from regulatory-driven carbon offset markets like those developing in California.

1. INTRODUCTION

Increasing attention is being paid to livestock, given the potential for shifts in management to have benefits for global climate as well as air and water quality (Smith et al. 2007a, 2007b; McCarl and Schneider 2001; Shindell et al. 2012; Hellwinckel and Phillips 2012). With increasing population and demand for food, particularly meat and dairy, strategies for mitigating livestock's impact on global climate change are urgently needed. Meeting mitigation objectives in the face of this demand and rising incomes will require dramatic shifts in human diets and in agricultural production systems that continue to expand (UNFAO 2000; Bennett and Balvanera 2007). However, where production is expanding little, the focus will likely be on shifting management of existing systems. Developed countries like the United States are likely to place greater emphasis on increasing these systems' efficiency rather than on developing new systems (Gerber et al. 2011).

Agriculture contributes approximately 6% of total greenhouse gas (GHG) emissions in the United States, and emissions from livestock production make up more than half of that total (U.S. EPA 2011). As shown in figure 1, enteric fermentation, which releases methane from animal digestion, is the largest contributor from livestock, followed by emissions from manure and grazing lands (U.S. EPA 2011). Emissions from crops produced as animal feed are also significant. Although included under cropland soils in figure 1, these emissions are not discussed in detail in this report, because they are covered in companion reports (Olander et al. 2011; Eagle et al. 2012).

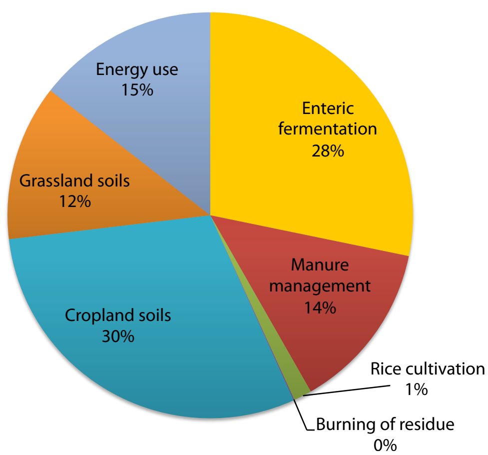

Figure 1. Gross agricultural GHG emissions, 2009
Note: Land use change and liming of soils are not included. *Source:* EPA 2011; USDA 2011a

Most of livestock's direct impact on global climate comes in the form of methane (CH_4), a potent greenhouse gas. Because methane is roughly 20 times more potent than carbon dioxide, small reductions in its output can have significant effects on overall emissions. Reductions in methane, and in black carbon, can greatly reduce predicted near-term temperature increases and climate change impacts. These reductions have the potential to keep global temperatures below the well-known 2°C threshold in the near term (Shindell et al. 2012). Livestock management is significant in methane mitigation efforts.

Within the United States, the second largest anthropogenic source of methane is enteric fermentation from livestock; in 2010, this source accounted for 70% of the annual CH_4 emissions associated with agricultural production systems (141.3 teragrams [Tg] CO_2e), which is equivalent to 21% of total U.S. annual anthropogenic CH_4 emissions (U.S. EPA 2012). The fifth largest CH_4 source is livestock manure management: 52 Tg CO_2e or 8% of total. Emissions from livestock manure in the United States have increased by roughly 56% since 1990 (fig. 2), mainly as a result of increasing use of liquid manure management systems, which have higher CH_4 emissions than other management methods (U.S. EPA 2011). Enteric emissions have decreased since 2007, because the population of beef, dairy cattle, and sheep has decreased slightly.[1] Livestock populations and production have held relatively steady, indicating a need to find ways to achieve emissions reductions without large-scale changes in the number of livestock (U.S. EPA 2011; USDA 2011a).

As shown in figure 3, beef, dairy, and swine systems are the largest contributors of greenhouse gases in the United States (U.S. EPA 2011) and are thus the focus of this report. Across these systems, numerous changes to feed or pasture management can enhance efficiencies and reduce emissions. Although activities targeted at enteric emissions and feed strategies (such as improved pasture management for beef cattle) may have a localized impact, their overall potential may be relatively small; much relevant research is at an early stage. In contrast, options for manure management have received greater study and are more established, and many, particularly options for methane capture and flaring and combustion for energy, can significantly reduce emissions. However, the capital, operation, and maintenance costs of altering manure handling systems can present a significant barrier that will require external investment and changes in energy sector policy and business models.

1. Although populations of horses, bison, mules, burros, and donkeys have increased significantly during this timeframe, they constitute a small portion of total emissions (U.S. EPA 2012).

Quantifying emissions reductions from livestock systems is most likely to be accomplished with various modeling approaches. At a project level, a variety of models are being tested and used. Carbon offset programs and corporate supply chain and certification programs are developing related yet somewhat different accounting approaches for tracking GHG emissions from livestock. Carbon offset protocols are designed to track a change in emissions from a specific project (or management change) relative to a pre-project baseline. Life-cycle assessments (LCAs), on the other hand, are used to assess changes in a product's production system or supply chain, to indicate emissions hot spots, and to provide a relative measure for emissions comparisons across supply chains, products, and time. LCA methods are in the early stages of application and as yet have little consistency. Livestock management, especially with regard to manure management, offers measurable and substantial CH_4 reduction potential. Moreover, many opportunities to reduce enteric emissions and improve land management may be worth pursuing.

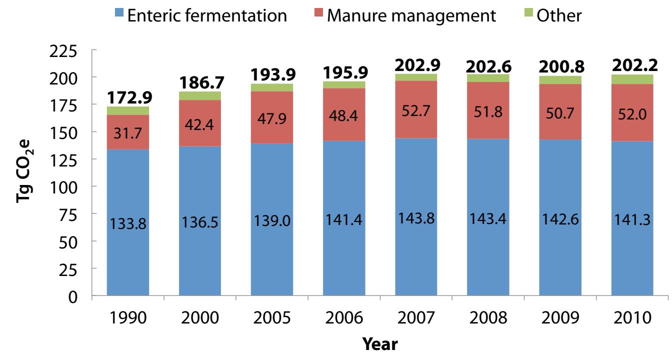

Figure 2. Sources of CH_4 emissions from U.S. agriculture
Source: U.S. EPA 2012

Although some management changes will increase production efficiency and thereby create overall cost savings, others will not. A handful of federal programs, such as the USDA's EQIP program, can help offset costs, but funding is insufficient for large-scale change. New markets for GHG offsets—both voluntary and in California's new cap-and-trade program—may also provide some compensation for adoption of new livestock management techniques.

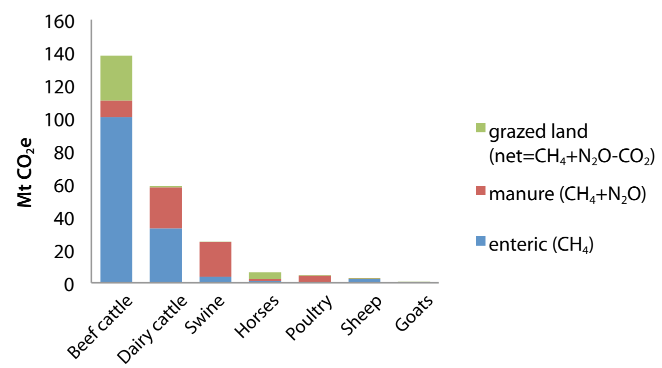

Figure 3. Greenhouse gas emissions by livestock source, 2008
Source: USDA 2011a

Corporate- and government-driven changes in supply chain management may also lead to increased demand for change in management practices (see Walmart 2010 and Executive Order 13514 for examples). Agricultural trade associations are developing programs to respond to these demands and to facilitate supply chain sustainability while keeping costs down (see Innovation Center for US Dairy 2010 for one example). If these trends continue, the agricultural sector may have a strong financial incentive to capture the value of reduced emissions.

2. DAIRY AND BEEF PRODUCTION

The major source of enteric CH_4 is beef cattle (72%), and within the beef cattle classification, the cow herd. Dairy cattle contribute an additional 23% (fig. 4). Sheep, goats, American bison and mules, burros, and donkeys account for an additional 1.2% of enteric emissions. Due in part to the reduced size of their symbiotic enteric microbial pool, horses and swine account for a total of 3.9% (U.S. EPA 2012) of enteric CH_4 emissions (286 gigagrams [Gg] CH_4 in 2012). The contributions of sheep, goats, American Bison, mules, burros, donkeys, horses, and swine have remained fairly constant since 2007 (U.S. EPA 2012). Wild ruminants are not included in the ruminant estimate due to their relatively low population numbers.

Nitrous oxide (N_2O) emissions from agriculture are primarily from soil (207.8 Tg CO_2e) and manure management (18.3 Tg CO_2e) (see fig. 5); a small amount comes from agricultural residue burning (0.1 Tg CO_2e) (U.S. EPA 2012). Nitrous oxide emissions from manure management include both direct emissions, from nitrogen cycling within manure and urine, and indirect emissions, which result from volatilization, runoff, and leaching during handling processes. Direct emissions from agriculture soil management are estimated to be 162.3 Tg CO_2e; they include N_2O emissions from manure deposited by grazing animals on grasslands and nitrogen from synthetic fertilizer, managed manure, and unharvested nitrogen residue. Total estimated pasture, range, and paddock manure N_2O emissions are 12.6% of the total. Total indirect N_2O emissions from soil are estimated to be 45.5 Tg CO_2e; the largest proportion (84%) is associated with croplands, and a small portion (13%) is associated with grasslands (U.S. EPA 2012). Livestock are also associated with a portion of the N_2O emissions resulting from the fertilization of crops used in animal feed. Apportionment of these emissions is not covered here.

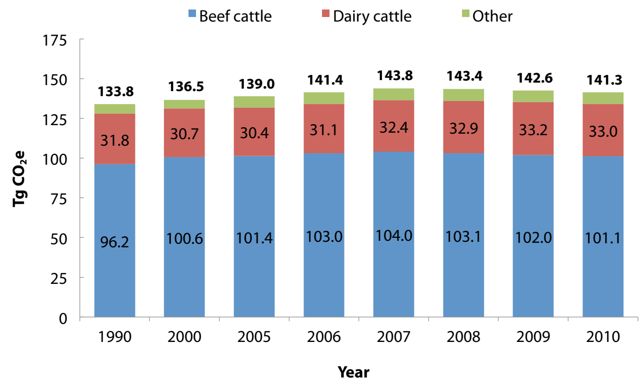

Figure 4. Distribution of enteric methane emissions
Source: U.S. EPA 2012

Dairy and Beef Production Systems

EPA inventory estimates indicate that 54% of the CH_4 production from ruminant systems is associated with the cow/calf enterprise for beef production, 23% of the total ruminant emissions are from dairy cattle, and the remainder of the emissions are from stocker and feedlot beef production systems (table 1) (U.S. EPA 2011). Methane emissions from enteric fermentation are dependent on diet quality; the highest emissions result from diets that include both forage and grain such as those fed to dairy cattle, and lowest emissions are from feedlot diets (high grain, low forage) (Johnson and Johnson 1995). Beef cows are responsible for the largest proportion of ruminant CH_4 emissions, primarily because of their relatively large numbers and not because they emit more CH_4 per animal relative to other ruminant livestock. In fact, dairy cows have significantly greater emissions per animal because of their diet (Johnson and Johnson 1995). Identification of those areas in which mitigation is possible hinges on an understanding of the diets for each group of animals and of typical production systems. Within each system, altered management may reduce emissions. Also critical is an examination of the entire livestock system to identify areas where efforts to reduce one greenhouse gas may result in unintended and undesirable increases in another.

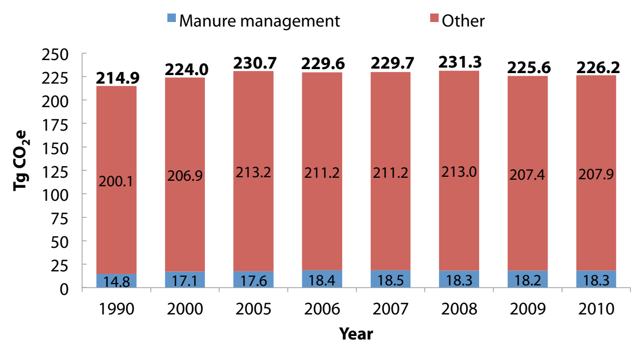

Figure 5. Nitrous oxide emissions from U.S. agriculture
Source: U.S. EPA 2012

Ruminant production systems vary by animal type (beef or dairy), size of operation, available feed resources, and region of the country. The following discussion is broadly representative of beef and dairy production systems, recognizing that large diversity in these systems exists in the U.S. Variability in these systems presents both opportunities for and barriers to adoption of GHG mitigation strategies (Field and Taylor 2008). Table 1 provides an overview of the phases of beef and dairy production and the categories of animals managed within each production system.

Economic conditions drive the population decisions made within a herd. For example, calves may go to a stocker system or directly to the feedlot depending on expected production costs and projected sale prices. Additionally, producers may maintain more replacement heifers than they need because there is a market for bred heifers. Decisions are made on the current and predicted economic situation and production goals. Any mitigation strategy that will enhance the economic situation of a given operation will receive serious attention.

Table 1. Phases of beef and dairy production

Animal class	
Beef cattle	
Beef cows	Cow-calf operations are found across the United States but are most numerous where pasture, range, or crop aftermath is available. The goal of these operations is an animal weighing 50% of its mature body weight at weaning (approximately 7 months of age). Cows may be bred for spring calving or fall calving depending on operation goals and may remain in herds for 10 or more years depending on productivity.
Bulls	Bulls are raised at a ratio of 1 bull to 25–35 cows. Bulls may be used as yearlings but are 2-year-olds when they are added to the herd.
Stocker	Stocker cattle are weaned calves that graze pasture or range for an additional 60–200 days and gain between 0.7 to 1.1 kg $hd^{-1}d^{-1}$. Stocker operations are present where forage supplies are inexpensive.
Replacement heifers	Replacement heifers are selected at weaning and their number is dependent on production goals. Generally, not more than 10%–15% of heifer calves are kept to be replacements. Heifers are usually bred at 12–15 months of age.
Feedlot	Feedlots may house as few as several hundred to more than 100,000 animals. These animals include beef cattle and dairy steers. The largest lots tend to be located in the Southern Plains. Beef cattle (steers and heifers) fed in feedlots are estimated to number 23 million annually (NASS 2011).
Dairy cattle	
Dairy cows (lactating)	Dairy cows enter the milking herd after calving for the first time at approximately 24 months of age and remain in the herd for the next 2 to 4 lactations. Diets are generally a mixture (total mixed ration, TMR) of high-quality forage and grain.
Dairy cows (dry)	Dairy cows are dried off approximately 60 days before the next calving. At that time, they are fed good-quality forage.
Dairy calves	Dairy calves are raised in calf hutches or barns for approximately 8 weeks until weaning and then raised as replacement heifers until calving at 24 months of age. Most heifers are kept and raised as replacements, whereas males will typically enter beef production as feedlot cattle.
Dairy heifers	Dairy heifers are weaned at 2 months of age. Heifers are bred at 12 to 14 months of age and enter the milking herd after calving. Heifers may be raised on farm or at specialized heifer-rearing enterprises and returned to the farm before calving.

Beef production demographics

Demographics

The 92 million U.S. beef cattle are found in cow/calf, stocker, and feedlot operations. The total number of cattle operations and the number of cattle have tended to decrease in the 2000s (USDA 2010) at the same time that the industry structure has changed. Industry statistics show increasing numbers of cattle found in relatively large operations. In the beef cow sector, 3% of the population is found in operations of 100 to 499 head (fig. 6). The nation's cow herd consists of approximately 30.8 million animals, both cows and replacement heifers, and is dispersed across the country (fig. 7), but approximately 16% of the population is located in Texas. The top 10 states, in descending order, are Texas, Oklahoma, Missouri, Nebraska, South Dakota, Kansas, Montana, Kentucky, Tennessee, and Florida. Together they account for 59% of the cow/calf numbers. The top states associated with the cattle-feeding industry (fig. 8) are Texas, Nebraska, Kansas, Colorado, Iowa, California, Oklahoma, Arizona, South Dakota,

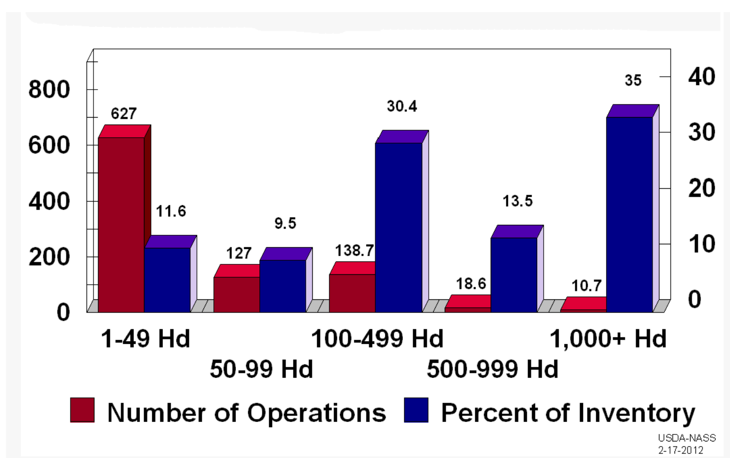

Figure 6. Beef cattle demographics
Source: NASS 2012

and Idaho which together make up 95% of the cattle on feed (NASS 2011).

Diet

Beef production systems are described in table 2 and are variable depending on the area of the country. In general, calves are weaned at 170 to 230 days of age (USDA APHIS NAHMS 2008) and may move to pasture (stocker) or go directly to the feedlot. Greenhouse gas emissions from the different production systems are associated with animal (enteric CH_4) or manure management. Table 2 identifies the types of diets fed to beef cattle at different stages of production. Cow/calf (including replacement heifers and bulls) and stocker cattle contribute most to enteric CH_4 emissions because of the forage content of their diet and their large proportion of the U.S. herd (Johnson and Johnson 1995). To date, options for mitigation of these emissions are few, but should they increase, the impact on total GHG emissions by the beef sector would be large. Additionally, any mitigation strategy available for the grazing cow herd would be usable by the stocker sector as well, enhancing the emissions impact. Because the diets of feedlot cattle contain large amounts of starch and limited roughage, enteric emissions are much reduced (from 6%–7% to 2%–3% of dietary intake energy). Feedlot diets also are variable in composition because regional byproducts are used extensively. The use of byproducts is important, because without cattle many of these products would be sent to the landfill to decompose. One of the more salient demonstrations of this process is evident with the expanded use of co-products (e.g., dried distillers grains with solubles) from the ethanol biofuel industry. In 2005, approximately 7.92 million metric tons of ethanol co-products were consumed by the U.S. cattle; by 2010, that figure had jumped to nearly 26 million metric tons (Renewable Fuels Association 2012). This consumption allows non-fossil fuel industries such as ethanol to remain economically viable.

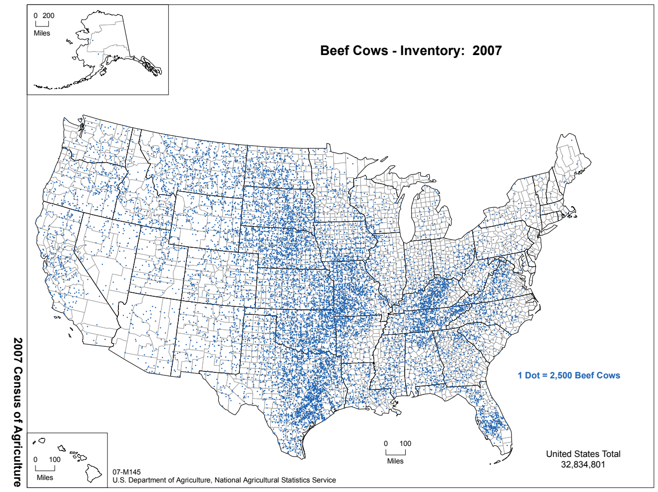

Figure 7. Location of cow/calf operations
Source: NASS 2010

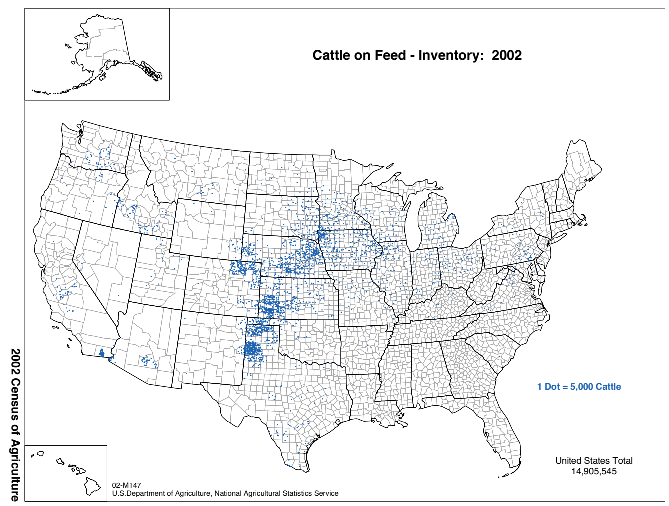

Figure 8. Location of the feedlot industry
Source: NASS 2002

Table 2. Beef cattle diets

Animal class	
Beef cows	Forage is the primary feedstuff, but when the pasture or range is insufficient to meet nutrient requirements, cows are supplemented with hay or grain (if prices are low). A recent examination by USDA (2010) of cow/calf feeding practices determined that 97% of cow/calf producers supplemented cows with additional forage (154 ± 7.0 days), 74% of producers supplemented with protein (173 ± 9.6 days), and 51% fed supplemental dietary energy (162 ± 12.7 days).
Bulls	The primary diet is forage, although some supplementation (hay or grain) may be added to the diet before the breeding season.
Stocker	Feedstuffs include improved perennial pasture, native range, annual crops such as wheat pasture, or crop residues such as wheat stubble or corn stalks.
Replacement heifers	Heifers are grazed or fed good-quality forage diets.
Feedlot	At the feedlot, calves are fed a high grain diet for 110 to 180 days until they are ready to harvest. In the southern plains, the diets predominantly consist of steam flaked corn, corn silage, and protein supplements. Feedlots in the northern plains feed greater amounts of corn co-products such as distiller's grains as well as dry rolled corn. Ionophores are included extensively in diets in conventional production systems.

Dairy production demographics

Demographics

The dairy industry in the United States consists of the milking herd, dry cows, and replacement heifers and calf rearing operations. In 2009, the milking herd was estimated to consist of approximately 9 million cows (NASS 2009), 55% in herds of more than 500 animals and 31% in herds of more than 2,000 animals (USDA 2011a). The dairy cattle population is found across the country, with population centers in California, Wisconsin, and the Northeast (fig. 9). Production has grown in the western and decreased in the southeastern United States (USDA 2007). The trend is fewer dairy operations with greater numbers of cattle. In 2007, the number of dairies was 51% lower than the number in 1991, but the number of cows was virtually unchanged (USDA 2007). Milk production per cow has increased 32.7% since 1991. In 2011, USDA estimated the United States had 4.4 million replacement heifers (USDA 2011a). Heifers may be raised on the farm or by heifer rearing facilities. As dairy operations increase in size, more heifers are being raised off-site in specialized operations, which made up 9.3% of dairies in 2007 (USDA NAHMS 2007).

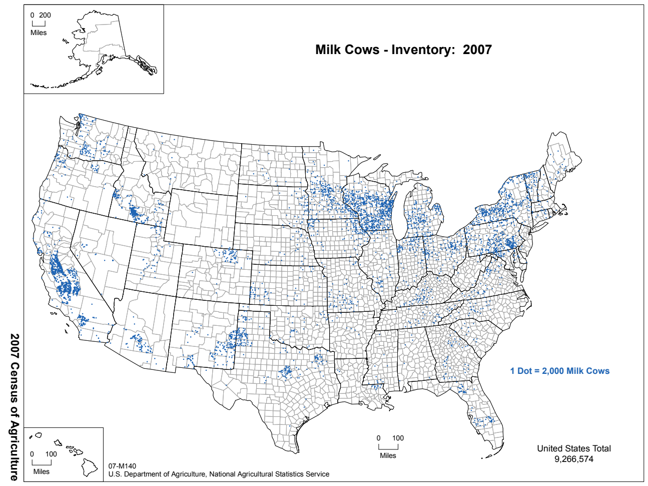

Figure 9. Location of the dairy industry
Source: NASS 2010

Like beef production systems, dairy production systems have management practices that create both opportunities for and barriers to incorporation of GHG mitigation strategies. The two areas in which GHG emissions arise from dairy systems are enteric CH_4 and manure management. Much of the research to reduce enteric emissions has focused on alterations in diet composition (Beauchemin et al. 2008). Table 3 describes typical types of diets fed to dairy cattle in the United States by animal classification and presents baseline conditions upon which mitigation strategies will be added.

Diet

Table 3. Dairy cattle diets

Animal class	
Dairy cows (lactating)	Diets are generally a mixture (total mixed ration, TMR) of high-quality forage and grain. Cows are managed in confinement feeding systems, dry-lot systems, or pasture-based systems and are fed diets that are 50% to 60% forage including silage, haylage, and hay. The balance of the diet consists of concentrates such as corn and other locally available ingredients such as soybean meal, cottonseed, or corn co-products. Diets are formulated as a TMR based on milk production level. Cows found in pasture-based systems will generally have the pasture diet supplemented with stored feeds when necessary to meet animal requirements. Approximately 33% of U.S. dairies rely on some pasture for part of the diet (USDA 2007). The USDA 2007 survey also indicated that approximately 27% of the dairies include ionophores in animal diets.
Dairy cows (dry)	Diets consist of good-quality forage.
Dairy heifers	Dairy heifers are fed good-quality forage and grain. Replacement heifers are fed diets balanced for growth rates of approximately 1 kg/d, which requires high-quality ingredients.
Dairy calves	Dairy calves are raised in calf hutches or barns and then raised as replacement heifers until calving. They are fed milk replacer, starter grain, and hay or other roughage.

Enteric Emissions

Overview

The direct sources of GHG emissions from ruminant production systems, as in other livestock production systems, are enteric fermentation and manure. Due to the inherent differences in gastrointestinal development, however, some aspects of these emissions are greater within ruminant production systems. Methane results from enteric fermentation and manure storage, whereas N_2O results from nitrification/denitrification processes on the ground or pen surface. Ammonia, although not a GHG, is another important emission associated with manure handling procedures, and it may lead to indirect formation of N_2O as it volatilizes and deposits in a different location.

The bulk of the GHG emissions from ruminant production systems are due to the natural release of CH_4 from the enteric fermentation within beef and dairy cows, bison, sheep, goats, and other ruminants. The enteric CH_4 is released primarily due to the action of methanogenic bacteria within the ruminant digestive system. Although most terrestrial animals possess methanogenic bacteria within their digestive systems, ruminant animals are unique in the symbiosis that they have with their digestive tract–associated bacteria. In most mammals, the majority of symbiotic bacteria are located within the hindgut (cecum and large intestine). The ruminant has a well-developed forestomach that comprises a large portion of its digestive tract. In fact, the contents of the rumen of a mature cow may account for as much as 20% of the total body weight of the animal. Therefore, because of the presence and fermentative capacity of this well-developed forestomach, ruminants emit much more CH_4 than non-ruminants.

Enteric and hindgut CH_4 emissions have been measured for decades using respiration calorimetry chambers in studies examining ruminant energy use. From these data, many equations have been developed to predict emissions (see “Measurement and Prediction of Methane Emissions from Livestock Systems” on page 33). The factors driving enteric CH_4 production are level of intake and diet chemical composition (Johnson and Johnson 1995). Table 4 illustrates the range of emissions measured from cattle fed different diets.

Table 4. Range of enteric methane emissions from dairy and beef cattle production systems

Animal type	Diet	CH_4 emissions range	References
Beef cows	Improved pastures; stockpiled forage; range fall and spring; Timothy pastures; Fescue pastures with intensive grazing management; grass pastures with intensive grazing management; rotational grazing, alfalfa, or grass; grazed continuously or rotationally with different stocking rates	115–273.3 g/hd/d	Westberg et al. 2001; Olsen et al. 1997; Pinares-Patiño et al. 2003; Pavao-Zuckerman et al. 1999; DeRamus et al. 2003; McCaughey, Wittenberg, and Corigan 1997, 1999
Dairy cattle (lactating)	Various	250–429 g/hd/d	Holter and Young 1992; Crutzen, Aselmann, and Seiler 1986; Johnson and Johnson 1995; McGinn et al. 2004; Sun et al. 2008; Loh et al. 2008; Beauchemin et al. 2008
Feedlot cattle	High grain	142–357 g/d	Johnson and Johnson 1995; Beauchemin et al. 2008; Loh et al. 2008; Hales, Cole, and MacDonald 2011

Although enteric fermentation is likely to produce significant amounts of CH_4, the conditions are unlikely to contribute to significant formation of nitrous oxide (N_2O). Recent work by Reynolds et al. (2010) detected no N_2O emissions when cattle were housed in respiration chambers equipped with N_2O analyzers.

Opportunities for mitigation

The effectiveness of various practices to alter enteric CH_4 depends on diet and management. The IPCC provided region-level estimates for the potential to reduce enteric CH_4 emissions while maintaining levels of productivity from various classes of livestock (Smith et al. 2007). In North America, it was assumed that improved feeding practices (improved roughage:concentrate ratio and balance of required nutrients in diets) could reduce total livestock enteric methane emissions by 16% in dairy cattle, 11% in beef cattle, and 4% in sheep. Specific agents and dietary additives (e.g., bST, ionophores, propionate precursors) are indicated to reduce enteric emissions of dairy cattle by 11%, beef cattle by 9%, and sheep by 0.4%. Improving inherent animal performance (including genetic selection) is estimated to reduce enteric methane emissions by 3% in dairy cattle, 3% in beef cattle, and 0.3% in sheep.

Viewing these potential reductions as additive is tempting, but the literature indicates that implementing multiple mitigation practices at once may reduce the effectiveness of any or all of the practices. Each enteric mitigation strategy must be evaluated within the context of the given production system (including dose and existing management practices) to achieve a relatively accurate estimate of the CH_4 emissions reduction in each class of livestock. The IPCC assessment (Smith et al. 2007) accounted for non-additivity by reducing the effectiveness of subsequent practices by 20%. An example of non-additivity might be altering the diet to increase propionate production by shifting the forage:concentrate ratio toward concentrate and adding additional propionate enhancers to the new diet. The impact of the propionate enhancers to CH_4 emissions in this scenario would be much reduced.

Enhanced animal production

Management strategies that increase animal productivity or improve efficiency of feed utilization can reduce CH_4 emissions per unit of product produced (g CH_4/kg beef or g CH_4/kg milk). Cooprider et al. (2011) conducted a simplified life-cycle analysis and found that CH_4 and N_2O emissions from cattle within a "natural" program (i.e., one without feed additives or growth-promoting agents such as hormonal implants, beta agonists, and ionophores) would produce quantities of CH_4 similar to those of cattle that received the performance-enhancing technologies on a daily basis. However, the "natural" cattle would require 42 additional days to reach the same final slaughter body weight (596 kg) due to reduced rate of gain. Therefore, the inefficiencies (or use of energy for purposes other than production of additional salable product) in energy use associated with maintenance would have increased over those 42 days, resulting in 1.1 kg greater CH_4 emissions per kg of body weight for the "natural" cattle than for those raised in the typical commercial system. Practices such as appropriate supplementation of nutrients that might be limiting (e.g., selenium, phosphorus, protein) result in performance enhancement and reduction in the amount of GHG produced by the animal production system per unit of product output (i.e., growth-promoting implants, ionophores, and so on). Other management practices to improve productivity will also reduce CH_4 emissions for the entire system. For example, changes in reproduction management that result in a greater number of pregnant cows will decrease the numbers of replacement heifers needed and thus reduce the total CH_4 emissions associated with milk or meat production (Capper 2011). Most mitigation options associated with reduced enteric methane are also associated with improvements in the efficiency of feed use by the animals.

Management practices that mitigate GHG emissions per unit of product produced in the beef or dairy industries are those that enhance efficiency of production and can be divided into three categories: improved diet digestibility, use of additives, and improved genetics of livestock.

Improved diet digestibility

Many reviews discuss dietary factors that may influence enteric CH_4 emissions (Johnson and Johnson 1995; Monteny, Bannink, and Chadwick 2006; Beauchemin et al. 2008; Eckard, Grainger, and de Klein 2010; Martin, Morgavi, and Doreau 2010). Improved digestibility of feed can increase the production/maintenance ratio of livestock and reduce the feed energy inputs required per unit of production, thereby reducing emissions of CH_4 per unit of animal product. Diet composition can also affect ruminal fermentation, altering the composition of volatile fatty acids produced, which affects CH_4 production, e.g., replacing sugars with starches in feed concentrates (Monteny, Bannink, and Chadwick 2006).

Benchaar, Pomar, and Chiquette (2001) modeled the effects of several dietary modifications on the enteric CH_4 production of a 500 kg dairy cow using a rumen digestion model. As shown in table 5, an example of modulating dietary digestibility, ammoniation disrupts the plant architecture and adds nitrogen, which increases digestibility and CH_4 production as a percent of dietary intake energy. Animal performance can be enhanced because of increased dietary energy, which can reduce the amount of CH_4 per product. Roughage preservation, forage species, and an increase in the concentrate:roughage ratio may all reduce the proportion of dietary intake energy lost as CH_4 by nearly 30%. However, management needs, such as animal longevity, milk quality, and prevention of dietary-induced disease, must be factored into decisions to adopt these types of changes. Additionally, the estimate of CH_4 reduction comes from a simulation model and does not reflect possibilities within ruminant production as a whole due to various logistical barriers, including economics and animal welfare. Current U.S. production systems (i.e., feedlots) might have the capacity to adopt a more extreme approach to the concentrate:roughage ratio that will result in a reduction in the proportion of dietary intake energy lost as CH_4. However, a producer managing a cow/calf operation would not find increasing the concentrate:roughage ratio modeled in Benchaar, Pomar, and Chiquette (2001) acceptable or possible due to cost, logistics, and concerns over the longevity of cows fed high-concentrate diets. There are limits to the quantity of concentrate feeds that can be safely included in a ruminant diet, and the leakage effects associated with the production of the grain may not yet be fully understood.

Leakage is the phenomenon through which efforts to reduce emissions in one place simply shift emissions to another location, sector, or land use where they remain uncontrolled or uncounted. The potential for leakage arises when the rules, regulations, and incentives for action affect only part of the potential pool of participants or emissions sources. If everything is counted, there is no leakage.

Table 5. Influence of various dietary strategies on enteric methane production in dairy cows using modeled simulations

Strategy	CH_4* (% of dietary energy intake)
Reduce methane	
Increase intake	-9 to -23%
Increase concentrate:roughage ratio	-31%
Use fibrous concentrate (beet pulp) rather than starch concentrate (barley)	-24%
Use rapidly (barley) rather than slowly (corn) degraded starch	-16%
Preserve roughage (dried vs. ensiled)	-32%
Process roughage	-21%
Increase methane emissions and animal performance	
Roughage maturity	+15%
Roughage species (legume vs. grass)	+28%
Ammoniation of straw	+500%
Supplement poor-quality roughage (straw)	+300%

Source: Benchaar et al. 2001.
* CH_4 emissions (losses) are generally expressed as a proportion of total dietary energy intake.

Lipid supplements

Numerous studies have demonstrated that supplemental fat may decrease enteric CH_4 emissions (Machmülller, Ossowski, and Kreuzer 2000; Jordan et al. 2006a,b,c), although possibly at the expense of reduced fiber digestion (McGinn et al. 2004). Beauchemin et al. (2008) indicated in a review that the production of enteric CH_4 decreased 5.6% for each 1% increase in added fat to the diet; Martin, Morgavi, and M. Doreau (2010) indicated that the reduction in enteric CH_4 production was 3.8% with each 1% increase in dietary fat. A meta-analysis by Moate et al. (2011) agrees with Martin, Morgavi, and Doreau (2010); it found a 0.79g CH_4/kg dry matter intake reduction for each 1% increase in fat. Additionally, Moate et al. (2011) determined that the type of fat fed to lactating dairy cows did not matter and found no indication that there was ruminal adaptation to the addition of fat through seven weeks of feeding. Significant negative impacts on the concentration milk fat and protein occurred, but the increase in milk production meant that fat and protein yield was unaffected.

Grainger and Beauchemin (2011) also conducted a meta-analysis on plant lipid supplementation and CH_4 emissions and determined cattle responded differently than sheep. Methane emissions were reduced 2.6-fold higher when sheep received increased dietary fat from plant sources. This analysis also found no difference in the form of added fat (oil versus seed), fatty acid balance in the diet, or plant lipid source. No conclusion could be reached regarding the persistency of CH_4 reduction, because the summarized studies did not report that information. On a practical level, these analyses have great value to formulation diets for dairy cattle or feedlot cattle, because the means of supplementation is straightforward. Supplementing grazing cattle is more difficult, but researchers are looking for high lipid-containing pasture plants and methods by which to add lipid to water. However, care must be taken with the addition of fat to reduce enteric CH_4 emissions, because ruminal microbes cannot tolerate high levels of dietary fat, and total dietary fat in the diet must be kept below 8% of the diet to avoid suppression of plant fiber digestion, which is counterproductive to grazing plant material and can cause substantial digestive upsets (Martin et al. 2008).

Feeding of co-product ingredients such as distiller's grains and other co-products of the milling and ethanol industries are one way in which fat is added to the diet. The impact on cattle CH_4 emissions from this practice has been measured; the results are variable and related to the total dietary lipid concentration. McGinn et al. (2009) measured a 25% to 30% reduction in enteric CH_4 with supplementation of distiller's grains, but Hales, Cole, and MacDonald (2011) found no effect. The reduction noted by McGinn et al. (2009) occurred when the diets containing distiller's grains had approximately 3% more fat than the control diets (due to the inherently greater concentration of fat in these co-products) whereas the diets used in the study by Hales, Cole, and MacDonald (2011) had similar concentrations of fat between the test and control diets.

Distiller's products are incorporated into feedlot and dairy diets and are increasingly being added to beef cow diets when protein supplementation is required. However, many of these products are high in phosphorus and protein, imposing limits on their dietary inclusion. When protein is fed in excess of animal requirements, increased NO_3 leaching, NH_3 emissions, and N_2O emissions from the pen surface and manure can occur.

Technologies to extract the lipid from distiller's co-products for use as a biofuel are being widely adopted. Over 40% of distillers products have had the lipid fraction removed. Removing the lipid that would have been added to the diet by inclusion of the distiller's co-products will decrease the observed CH_4 suppression effect (Renewable Fuels Association 2012). However, both the dairy and feedlot segments are likely to add other lipid-containing ingredients to maintain diets' energy density. If the lipid addition is plant lipid, a CH_4 suppression effect is likely.

Pasture management

Studies find little to no evidence that management-intensive grazing (e.g., rotational grazing, improved pastures, fertilization) will reduce enteric CH_4 emissions per animal (Leng 1991; McCrabb, Kurihara, and Hunter 1998, Pavao-Zuckerman et al. 1999; DeRamus et al. 2003; Pinares-Patiño, Baumont, and Martin 2003). However, there is ample evidence that animal productivity is increased, both the maintenance of cows' body condition and calves and stockers' enhanced growth rate, which can reduce CH_4 emissions per unit of animal product. There is also evidence that pasture management increases carbon sequestration in soils, which is not the case for the management of extensively grazed rangelands (Eagle et al. 2012). One caveat is that management-intensive grazing may require irrigation or fertilizer use. Both practices can lead to increased GHG emissions: N_2O from fertilizer use and CO_2 from fuel use for fertilizer production, spreading, and irrigation. To address the increased GHG emissions with fertilizer use, New Zealand supports application of N_2O inhibitors to suppress emissions. This technology is not commonly used in the United States.

> **Management-intensive grazing** describes grazing systems in which animals graze a small portion of a pasture while allowing the other sections to rest and recover. The intent is to manipulate the quantity and quality of the forage base to promote animal performance and plant community health.

Additives/pharmaceuticals

Many potential dietary additives and agents have been associated with a reduction in enteric methane emission. Common issues with several of these additives are inconsistent reductions in enteric CH_4 and a transitory reduction of CH_4 production in the rumen as the rumen microbes adapt to the agent. Several of the techniques that may reduce enteric CH_4 emissions may also limit the acceptability of the final products for several markets. For example, antibiotics and the ionophore monensin may have potential to reduce CH_4, but raise concern about the proliferation of antibiotic-resistant bacteria. Additionally, several markets (including the European Union) ban the use of several of these additives in the production of livestock products.

Ionophores

Ionophores (such as lasolocid or monensin) are one of the most common additives to be associated with a reduction in enteric CH_4 production, which may reduce methane emissions by 10% to 25% (Benz and Johnson 1982; Van Nevel and Demeyer 1996; McGinn et al. 2004; Tedeschi, Fox, and Tylutki 2003). The CH_4 suppression effect does not appear to be permanent with cattle-fed feedlot diets (Rumpler, Johnson, and Bates 1986; Guan et al. 2006) but may be permanent with mixed diets such as those fed to dairy cattle. Odongo et al. (2007) reported that monensin fed to dairy cattle decreased enteric CH_4 by 7% to 9% for up to six months. This same study found a reduction in milk fat (9%) and milk protein (4%), which would have serious implications for milk marketing. Ionophores do increase animal efficiency, because they provide the same or enhanced performance with decreased feed intake. Ionophores may not have a direct effect to suppress CH_4, but they may have a role when the goal is reduced CH_4/unit of milk or meat produced.

Halogenated compounds

Halogenated compounds are highly reactive and have detrimental impacts on biological organisms such as the ruminal bacteria. Halogens include fluorine, chlorine, bromine, and iodide. Halogenated compounds are those that contain a halogen (e.g., bromochloromethane [BCM] and chloralhydrate). These compounds may reduce enteric CH_4 emissions by inhibiting the formation of CH_4 by methanogenic bacteria populations, but they may also decrease feed intake and animal performance (Johnson 1972). Studies examining levels of BCM added to feedlot diets found that CH_4 emissions were reduced 57% to 91% as the dose rate increased (Tompkins and Hunter 2004). However, the suppression effect may be transitory (Wolin, Wolf, and Wolin 1964; Van Nevel and Demeyer 1995). The acceptability of feeding halogenated compounds to animals is open to question and, therefore, these compounds are not thought to be a viable mitigation option (Eckard, Grainger, and de Klein 2010)

Plant compounds

There is a growing body of literature evaluating the role of various compounds from plants such as saponins (Lila et al. 2003), essential oils, and tannins (Pinares-Patiño et al. 2003) in reducing enteric CH_4. To date, information regarding the effectiveness of many of these compounds is limited and conflicting. Saponins are glycosides that have a direct effect on ruminal microbes by binding to microbial membranes, resulting in cell death. Goel and Makkar (2012) reviewed the available information regarding saponins and found as many reports indicating CH_4 suppression with saponin feeding as reports indicating no effect. Additionally, several reports indicated a reduction in diet digestibility, which is undesirable. Research is needed on the conditions under which saponins' addition to diets may be effective.

Essential oils can be used to suppress methane. The oils examined to date include garlic oil (allicin and diallyl disulfide), rhubarb, and frangula. These compounds are antimicrobials that interact with lipid membranes of the bacteria, in particular gram-positive bacteria (Jouany and Morgavi 2007), and that therefore can affect the ruminal fermentation.

There are hydrolysable tannins (toxic to the host animal) and condensed tannins. Condensed tannins in the diet have reduced CH_4 emissions from 0% to 30% depending on the study (Martin et al. 2010) and are thought to work by decreasing diet digestibility or directly affecting methanogenic bacteria. Several groups have investigated the use of tannins in diets to reduce N_2O emissions. Because tannins bind to proteins in the rumen and can carry the protein through the intestine (reducing digestibility and absorption), more nitrogen is found in the feces than in the urine, thus reducing N_2O emissions and slowing soil N metabolism (Eckard, Grainger, and de Klein 2010). The efficacy, persistence, and toxicity of tannins as dietary supplements will require additional research (Bayat and Shingfield 2012).

Nitrate and sulfate addition to the rumen can successfully compete for H_2 in the rumen, making it less available to methanogens. Ruminal metabolism of nitrate consists of reduction to nitrite and then to ammonia. Ruminal bacteria preferentially use ammonia for protein synthesis, which means supplementation with nitrate could enhance the efficiency of microbial protein synthesis while simultaneously decreasing CH_4. Unfortunately, accumulation of nitrite can cause potentially deadly nitrite toxicity. The ruminal microflora can adapt to high nitrate levels so nitrate supplementation would require careful management. Some research conducted in Japan noted that if L-cysteine or galactooligosaccharides were fed with nitrate, nitrite accumulation would lessen, thus reducing the potential toxicity (Takahashi 2011). There is also evidence that the addition of sulfate to a nitrate supplement results in additional suppression of CH_4 production (van Zijderveld et al. 2010). Farm managers may be able to use technology that limits intake of supplements for pasture cattle to effectively deliver nitrate supplements. Given the feeding risk involved in this strategy, adoption is unlikely to be easy.

Probiotics and organic acids

Probiotics, which may stimulate the growth of preferred populations, may also have a relatively small impact on enteric CH_4 (McGinn et al. 2004), but there is limited literature regarding this effect. Fumaric and malic acid may serve as propionic acid precursors to reduce methane formation by serving as an alternate H sink. These compounds must be fed at high doses to achieve a reduction in CH_4 (Newbold et al. 2005), making them an unlikely mitigation strategy.

Vaccination

Vaccination against methanogens may also reduce enteric production of methane (Wright et al. 2004), yet study results have been inconsistent (Eckard, Grainger, and de Klein 2010). The early research, conducted in Australia, reported CH_4 emissions reductions of 7.7%, but these results were not repeatable in subsequent studies—not surprisingly, given the complexity of the ruminal microbial population and alterations that may exist within the methanogen population when animals are fed different types of diets (Wright et al. 2004). Newer technologies using bacteriophages to directly inhibit methanogens or selectively enrich the rumen for other H_2-using bacteria (e.g., acetogens) may prove more effective, but this research is in the early stage (McAllister and Newbold 2008). Another longer-term avenue of research is the potential use of bactericins, naturally produced antimicrobial compounds produced by one species of bacteria to inhibit a competitor, from methanogenic bacteria to inhibit or control ruminal methanogens (Cottle et al. 2011).

Improved genetics

Selection for animals who emit less methane or those who emit less CH_4/kg DMI has been suggested as a method by which CH_4 emissions could be reduced (Hegarty et al. 2007). At this time, however, no methods for screening large populations have been validated. Several research groups are working on a screening technique. Schemes for selection of related traits may have more success. Selection of animals for improved productivity (e.g., growth, milk) relevant to the production system will reduce GHG emissions per unit of animal product. In addition, many have suggested that selection for improved feed efficiency (reduced feed intake per unit of gain) will also reduce ruminal enteric CH_4 production (Hegarty et al. 2007; Nkrumah et al. 2006). An animal that eats less and gains the same (or greater) weight than an animal that eats more will be alive fewer days. Most reports have relied on predictions of emissions reductions from models rather than on actual measurements. Hegarty et al. (2007) found decreased CH_4 emissions when animals were selected for improved residual feed intake (RFI, a feed efficiency trait). They also found that RFI explained only a small amount of the variation observed in CH_4 production and proposed the existence of a high genotype by diet interaction, which would decrease the effectiveness of single-trait selection for RFI to reduce CH_4. Furthermore, Jones et al. (2011) found that Angus cows selected for improved feed efficiency had lower CH_4 emissions compared with their inefficient counterparts when fed high-quality forage. However, no differences were detected when animals were fed lower-quality forage. Thus, more research is needed to understand how diet may interact with the RFI trait such that CH_4 emissions are not always reduced. Additionally, the observed effect or lack of effect on CH_4 emissions might be dependent on stage of production (lactation, dry, open, etc.) and type of diet (Jones et al. 2011). Freetly and Brown-Brandl (2011) present evidence that indicates that CH_4 emissions do not correlate with feed efficiency but are more closely related to less food intake. Therefore, selection for feed efficiency may not necessarily decrease CH_4 emissions per unit of feed consumed, but it will decrease the amount of feed needed to produce a unit of milk or meat, thus decreasing the amount of protein needed (and perhaps the amount of N lost to the environment). Research programs are examining polymorphisms in the bovine genome associated with animal performance, feed efficiency, and, to a limited extent, CH_4 production. Breeding systems will eventually incorporate these data, but the technology will not be sufficiently inexpensive for routine use for several years.

Changing production systems

In general, adopting more efficient production systems will help to mitigate all greenhouse gases, by allowing for production of the same quantity of products to support human needs, while reducing inputs (i.e., substrate for potential GHG production). Reviews have been conducted of the impact of the improved efficiency on overall environmental impact in both dairy cattle (Capper, Cady, and Bauman 2009) and beef cattle (Capper 2011). This paper focuses on the shorter-term changes within existing production systems rather than full-scale shifts in those systems (e.g., completely altering housing structures, manure management systems, and so on), given that such shifts are less likely to be efficient or fiscally feasible in the US where new production is coming from efficiency rather than the construction of new facilities and farms.

Critical research needs

Understanding of enteric CH_4 production and potential mitigation practices, while generally well developed, is hampered by lack of data in several areas. Methods for measuring feed intake by grazing animals are inadequate, making evaluation of mitigation strategies and modeling imprecise. Once intake is known, models and prediction equations for enteric CH_4 can be developed to examine the impact of a genetic or management strategy deployed to reduce GHG emissions in extensive systems. Understanding this impact is critical, given that such a large proportion of the cattle population exists within a pastoral or range setting. Concomitant changes in animal production (or lack thereof) associated with enteric CH_4 mitigation practices also need to be studied so that GHG mitigation potential can be accurately estimated. Lastly, greater understanding of the additivity (or non-additivity) of mitigation practices is needed. The relationship among mitigation activities is likely much more complex than the simple static reduction in additivity proposed by Smith et al. (2007), but to what extent is unknown.

Manure Management Emissions

Manure management

The biotransformations occurring in manure are a source of GHGs and may represent an area in which management may be able to mitigate emissions (table 6). Within the United States, manure management contributed approximately 49.5 Tg CO_2 equivalents from CH_4 production (fig. 3) and 17.9 Tg CO_2 equivalents from N_2O in 2009 (figs. 5 and 6). This 67.4 total Tg CO_2 equivalents from manure management therefore accounted for approximately 1.02% of total U.S. GHG emissions in 2009.

Table 6. Manure handling practices

System	Practice	GHGs
Beef		
Cow-calf; stocker	Pasture deposition	N_2O
Feedlot	Stockpile, compost, spread, seasonal catch basins, lagoons	CH_4, N_2O
Dairy		
Grazing	Pasture deposition	N_2O
Confinement	Stockpile, compost, spread, solids separation, anaerobic digester, lagoons	CH_4, N_2O

Nitrous oxide is produced from nitrification or denitrification of manure that is deposited on the ground or on a pen surface. Grazing animals deposit manure on pasture and rangelands, but GHG emissions associated with this activity are not available for the United States. Factors that influence the amount of N_2O emissions from grazing lands include plants, soil type (C/N ratio, pH, management, N application rate, timing), and environmental conditions (including moisture and ambient temperature). Additionally, animal and plant management techniques (stocking density, animal type and productivity expectations, plant type and composition) can affect N_2O emissions (Luo et al. 2010). Owens, Edwards, and Van Keuren (1989) found no nitrogen losses from low-input grazing (low stocking density, no fertilization), but as stocking density increases or as lands are fertilized, these losses increase (Jarvis, Wilkins, and Pain 1996). Management of any or all of these inputs can reduce N_2O emissions. For example, careful timing and rate of fertilizer application, coupled with grazing timing and stocking density, can reduce emissions (Pakrou and Dillon 2000).

Much of the recent work regarding N_2O emissions from manure deposition from grazing cattle are from dairy systems in New Zealand, where management-intensive grazing systems are used. Estimates of N_2O emissions from grazing lands are highly variable. The amount of N_2O-N loss from urine N deposited on pasture has ranged from 0.1% to 3% (Vermoesen, van Cleemput, and H.G. Hoffman 1997; Clough et al. 1996). Estimates of N_2O-N loss from feces ranged from 0.4% to 0.53% (Felssa et al. 1996; Yamulki, Jarvis, and Owen 1998). The emissions rates found in New Zealand are unlikely to be relevant to U.S. production, but the factors associated with N_2O emissions are those that should be considered for U.S. intensive grazing systems.

In feedlots some of the manure and soil from the pen surface may be used to create a mound to allow animals to have a dry place to lie down after rain or snow. In most feed yards, the pen surface is scraped and manure removed and stacked before and after sale of the cattle. In areas in which rain or snow is considerable, run-off-holding ponds with settling basins are common. EPA (2011) estimates that less than 1.5% of feedlots have liquid/slurry systems. Handling systems will result in differential emissions due to varying environmental conditions and management for odor and ammonia. In manure management, emissions tradeoffs are inevitable because of the shift between dry and liquid management systems.

Manure in dairy dry-lot housing systems is managed much like that in a beef feedlot, where the pen surface is scraped and manure and soil removed to stockpiles or compost facilities or spread directly onto adjacent farm land. Tie stall dairy facilities have gutters where manure is collected and transported to spreaders or a manure stack. Free stall barns are scraped and the manure is directed to a solids separator before the liquid flows to a lagoon or digester or sometimes directly applied to a field using a honey wagon. Solids (fiber) that are separated from the liquid waste stream will be stacked, and when field conditions are right, they are spread on cropland using a manure spreader. Dairies also use manure management systems such as covered run-off or collection basins and compost systems to reduce GHG emissions. EPA (2011) identifies manure handling on dairies to be direct deposit on pasture (primarily in southern states), daily spread (primarily in the eastern states), solid storage, liquid/slurry, anaerobic lagoons, and (less frequently), deep pit storage.

Leytem et al. (2011) characterized the emissions of CH_4, CO_2, NH_3, and N_2O from an open lot dairy in southern Idaho that also contained a compost facility and wastewater pond. Emissions varied considerably across all four seasons, but the highest emissions rates for all measured gases occurred in the spring and fall, and the lowest emissions rates occurred in the winter. The data for the open lot are reported per cow, with average emissions of 0.49 kg CH_4, 28.1 kg CO_2, and 0.01 kg N_2O. These emissions include both those associated with the cow (CH_4 and CO_2) and those associated with manure (CH_4, CO_2, and N_2O). Emissions from the wastewater pond had a diurnal pattern; the highest concentrations of CH_4 occur during the day (the annual average was 103 g $CH_4/m^2/d$) and lowest N_2O emissions, 0.49 g $N_2O/m^2/d$. The compost yard had CH_4 emissions ranging from 476 to 3,522 kg/d. Emissions were highest in the spring. The highest N_2O emissions (267 kg/d) for the compost yard also occurred in the spring; the annual range was 12 to 267 kg/d. The whole enterprise emitted, on average, 0.044 kg/d CH_4, 0.94 kg/d CO_2, and <.001 kg/d N_2O per kg of milk/d.

Treatment of slurry before field application can affect GHG emissions after application. Slurry that has been through a separator produced half the GHG emissions, and anaerobic digestor slurry application produced 60% fewer GHG emissions than untreated slurry (Amon et al. 2006). The few assessments of N_2O emissions reductions from management of manure application in North America estimate reductions ranging from 0.4 t CO_2e/ha/yr to 1.2 t CO_2e/ha/yr (Eagle et al. 2012) (table 7). However, the most promising change to management may be adjustments in commercial fertilizer application rates to account for the N added to manure. Estimates from USDA (ERS 2009) suggest that nearly 40% of farmers do not make these adjustments.

Table 7. Estimate of N_2O emissions reductions from improved manure application management

Citation	Region	Comments or caveats	Information source	Potential (t CO_2e ha^{-1} yr^{-1})
Paustian et al. (2004)	U.S. general	General estimate for improved "waste" disposition, 10% reduction in emissions	Expert estimate	1.17
Pork Technical Working Group (2005)	Canada	Apply to dry rather than wet areas, 50% reduction in N_2O emissions	Expert estimate	0.59
Gregorich et al. (2005)	Canada	Apply solid rather than liquid manure, review of 5 studies	Review, no individual data	0.86
Rochette et al. (2000)	Canada	Apply lower rate of pig slurry, reduces % N denitrified from 1.65% to 1.23%	Field study	1.22

Source: Eagle et al. 2011.

GHG emissions from the different manure handling systems used by dairies or feedlots were investigated by Pattey, Trzinski, and Desjardins (2005). Emissions of CH_4, N_2O, and CO_2 were measured from slurry, stockpile, and compost (passively aerated) from beef and dairy operations over the summer. Over the three-month experimental period, total GHG emissions for the beef manure were 51 g CO_2e/kg dry matter (DM) for the compost, 76 g CO_2e/kg DM for the stockpile, and 230 g CO_2e/kg DM for the slurry. A similar response was observed for the dairy manure emissions: 207 g CO_2e/kg DM for the compost, 301 g CO_2e/kg DM for the stockpile, and 397 g CO_2e/kg DM for the slurry. The authors went further and calculated the potential reductions in GHG emissions from manure if all of it was composted rather than stockpiled or stored as slurry and estimated that a 0.7 Tg CO_2e/yr reduction could be achieved. If all of the manure was handled as slurry and the CH_4 was collected and used for energy, a 1.08 Tg CO_2e/yr reduction could be achieved. This study needs to be continued across the seasons to provide information to managers about the interaction of manure storage options and environmental temperatures.

Differences in composting techniques also result in differential GHG emissions. Passive aeration systems are those in which the manure sits on perforated pipes through which air is introduced. Active aeration systems are those in which the compost is turned several times in the windrow. Hao et al. (2001) measured GHG emissions from feedlot manure with both of these systems and found emissions of 240.2 kg CO_2 Eq/Mg for the passive aeration and 401.4 kg CO_2 Eq/Mg for the active aeration. Some composting systems include a biofilter for odor reduction. Biofilters are effective for scrubbing the airstream of NH_3; they may slightly reduce or have no effect on CH_4, but they generally result in increases in N_2O emissions (Amlinger, Peyr, and Cuhls 2008).

Opportunities for mitigation

Although the type of manure management system is a major factor controlling the conversion of volatile solids into methane and the conversion of manure N into N_2O, climate (especially ambient temperature) can be an equal, if not greater, influence on the emissions of GHGs from manure for some systems. Methane conversion factors for the various management systems within the United States range from 0% to 0.75% of volatile solids, and the N_2O formation rates range from 0 to 0.1 kg N_2O/kg N in the manure, depending on the type of manure management system in use and local climate (especially ambient temperature) (EPA 2011). Therefore, the scale of the impact of modifying existing systems will be highly dependent on the type of manure management system in use in a given operation. Modifying manure management would be futile in systems with very low rates of GHG emissions, but improvements are possible for those systems that have greater rates of GHG emissions production.

Modifications of existing systems

A variety of manure treatments have been proposed to reduce GHG emissions from animal waste, including cooling manure, altering manure pH, compacting solid manure to reduce O_2, and frequent spreading of manure.

Manure cooling to less than 10°C, can lower overall microbial activity and therefore both CH_4 and N_2O emissions. Although this technique has been proven on a bench-top scale, it would likely be unfeasible at a commercial scale to provide the refrigeration capacity to cool the manure of a large CAFO (Sommer, Petersen, and Møller 2004).

Altering manure pH may inhibit overall microbial activity and hence both CH_4 and N_2O emissions, but the resource requirements (both financial and physical) would likely preclude this option from being feasible at this time (Berg 2003).

Compaction to denitrify manure all the way to N_2 might be an option to help reduce nitrous oxide. However, this process is likely to increase methane formation.

Frequent spreading of manure, if feasible, may reduce methane emissions that occur during the storage period but may result in additional emissions occurring at application time (e.g., NH_3 emissions that comprise a source for indirect N_2O emissions from manure). Little identifiable research data has examined this tradeoff.

Flaring of methane or use for energy would ultimately convert the CH_4 to CO_2, reducing global warming potential by 21 times. To help offset the installation, maintenance, and construction costs of the facilities required for this method, most producers sell the electricity generated to a local electrical grid and to acquire the associated renewable energy certificates (RECs) and carbon credits. Without the ability to sell electricity or obtain credits, costs prohibit producers from adopting this methodology.

Avoiding losses of gaseous N and leaching/runoff from stored manure will reduce off-site (indirect) N_2O emissions. Additionally, dietary nitrogen content can alter both urinary and fecal nitrogen content (Archibeque et al. 2007), which may contribute to N_2O production. However, data to properly estimate the potential impact of this form of management on subsequent N_2O emissions are lacking. For example, nitrous oxide formation is dependent on the presence of oxygen (Zumft 1997), whereas methanogenesis is an anaerobic process (Johnson and Johnson 1995). Therefore the formation of these gasses (N_2O and CH_4) under aerobic/anaerobic conditions are diametrically opposed where the reduced emissions from one gas can be offset by the increased emissions of the other.

Shifting management systems

Differences in manure handling systems affect the environmental factors that drive GHG formation in manure. IPCC (2006) indicates that the CH_4 conversion factors will range from 0% to 100% conversion of volatile solids (see table 8). The amount of CH_4 generated by a specific manure management system is affected by the extent of anaerobic conditions, temperature, and the time that organic material is held within the system. Essentially 0% of volatile solids are converted to CH_4 in aerobic systems, but as much as 80% are converted in highly anaerobic systems, such as those that use deep bedding or anaerobic lagoons. This variation illustrates how important selection of a manure management system is for mitigating CH_4 production from manure. In management systems with low CH_4 conversion factors (i.e., dry lot, daily spread, pasture), system modification is unlikely to have a significant impact on GHG emissions from manure.

Several logistic factors (such as climate, topography, necessary land, and accessibility to an electrical grid that will accept electricity) may compel a producer to utilize a given manure management system. The infrastructure and financial inputs required for the system may preclude the option of changing the system to mitigate GHG emissions from manure.

Table 8. Methane conversion factors for manure management systems, expressed as a percent of volatile solids excretion

System		Methane conversion factors by average annual temperature (°C)								
		Cool			Temperate			Warm		
		≤10	12	14	15	20	25	26	27	≥28
Pasture/range/paddock		1.0%			1.5%			2.0%		
Daily spread		0.1%			0.5%			1.0%		
Solid storage		2.0%			4.0%			5.0%		
Dry lot		1.0%			1.5%			2.0%		
Liquid/slurry	With natural crust cover	10%	13%	15%	17%	26%	41%	44%	48%	50%
	No natural crust cover	17%	20%	25%	27%	42%	65%	71%	78%	80%
Uncovered anaerobic lagoon		66%	70%	73%	74%	78%	79%	79%	80%	80%
Pit storage below animal confinements	< 1 month	3%			3%			30%		
	> 1 month	17%	20%	25%	27%	42%	65%	71%	78%	80%
Anaerobic digester		0%–100%			0%–100%			0%–100%		
Burned for fuel		10%			10%			10%		
Cattle deep bedding	< 1 month	3%			3%			30%		
	> 1 month	17%	20%	25%	27%	42%	65%	71%	78%	80%
Composting: in vessel		0.5%			0.5%			0.5%		
Composting: static pile		0.5%			0.5%			0.5%		
Composting: windrow		0.5%			1.0%			1.5%		
Aerobic treatment		0%			0%			0%		

Source: Adapted from table 10.17, IPCC 2006.

Critical research needs

Accurate estimation of the impact of management shifts on GHG production from manure is hampered by lack of understanding in three areas. First, how do manure-handling activities and myriad environmental conditions interact to influence GHG production from manure? Researchers must develop recommendations for optimal manure handling times (based on climate conditions) and for managerial decisions regarding stockpiles, composting, and spreading. Second, how do other mitigation activities influence GHG production from manure? This knowledge is critical to prevent leakage effects (e.g., reduction of GHG emissions from enteric fermentation but production of an equivalent amount of these emissions from manure). Third, what are the impacts of the linkages between dietary composition changes and manure management systems on overall GHG production from manure?

Mitigation Options with the Greatest Likelihood of Impact on Emissions from Beef and Dairy Production

In 2007, manure management was associated with 44 Tg CO_2e from CH_4 emissions and 14 Tg CO_2e from N_2O emissions (U.S. EPA 2009). Although CH_4 emissions could be reduced to negligible amounts with the construction of CH_4 digesters or systems that promote aeration of manure, the substantial infrastructure and maintenance costs of doing so would have to be offset through a combination of electricity and REC production, farm bill programs, and carbon offset markets. Nitrous oxide management may be more feasible, yet even a 10% reduction in N_2O emissions through improvements in manure management would equate to a relatively modest 1.4 Tg CO_2e reduction. Within manure management systems, dairy cattle, swine, and beef cattle produce 18.1 Tg CO_2e, 19.7 Tg CO_2e, and 2.4 Tg CO_2e manure CH_4, respectively, equating to 1.34 Gg CO_2e/1000, 0.303 Gg CO_2e/1000, and 0.036 Gg CO_2e/1000, respectively. Given the variation in CH_4 production associated with the various manure handling practices, it is theoretically, if not realistically, possible to remove almost all of the 44 Tg CO_2e that results from CH_4 emissions. Adoption of practices that capture the CH_4 or prevent it from forming must be balanced with the realization that aerobic conditions will enhance N_2O release through increased surface agitation in and air flow through the system. However, typically only 1% or less of manure N will be converted to N_2O. The conversion rate for some practices, especially actively mixed deed bedding and intensive windrow composting, can be as high as 7% to 10% (IPCC 2006).

Table 9 provides a qualitative summary and a comparison of the reviewed mitigation practices. The action category indicates whether the management action

1. is *ready* for integration into programs and protocols as a mitigation practice and is of *high* or *moderate priority* given its potential;
2. is likely to have significant mitigation potential but is supported by little research, making it a *research priority*;
3. appears to have low mitigation potential or significant implementation barriers and thus is a *low priority* for research or action; or
4. is supported by too little research to make a recommendation and is therefore *uncertain*.

Table 9. Summary of beef and dairy mitigation practices, based on the opinions of expert authors

	Mitigation potential	Amount of research	Expert confidence	Potential expense	Action category
Grazing systems changes					
Change from traditional pasture grazing to managed pasture	Some soil C benefits and efficiency of animal production	Low	Moderate	Moderate	Ready High priority
Change from traditional pasture grazing to intensive feedlot system	Will likely produce ~50% reduction in enteric CH_4, but more research needed on leakage issues	Significant	Significant	Significant	Research priority
Feeding strategies					
Lipid supplements	Unclear	Moderate	Significant	Low	Research priority
Intake modification/ measurement in association with reduced CH_4 production	Unclear	Low	Low	Unknown	High priority
Ionophores	10% to 25% reduction in CH_4, but duration may be limited*	Significant	Significant	Low	Research priority
Halogenated compounds	Low	Low	Moderate	Very high	Low priority
Plant compounds	Unclear	Low	Low	High at effective doses	Research priority
Probiotics and organic acids	Unclear	Low	Low	High at effective doses	Moderate research priority
Vaccination	Unclear	Low	Low	High	Low priority
Improved genetics	Unclear	Moderate	Low	High (long-term investment 10 to 20 years)	Research priority
Manure management					
Manure cooling of 10°C	Unknown	Low	Low	High	Low priority
Altering manure pH	Unknown	Low	Low	High	Low priority
Compaction	Theoretically high, but little data	Low	Moderate	Low	Research priority
Frequent spreading	Unknown	Low	Low	Low	Research priority
Methane use for energy – Methane digesters	High	Moderate	High	Very high	Ready – high priority and research priority
Manure aeration	High	Moderate	Moderate	High	Research priority

* Less is known about the potential to "cycle" the ionophore (e.g., 6 weeks on, 6 weeks off).

3. SWINE PRODUCTION

Direct emissions of GHGs from animal production in the United States have been estimated to be 203 million tonnes (Mt) CO_2e (carbon dioxide equivalents), or about 2.9% of total U.S. emissions (6,957 Mt CO_2e) in 2008 (CAST 2011a). Swine production comprises only a small portion of U.S. emissions; swine, sheep, poultry, and goat production, combined, represent less than 5% of U.S. enteric methane emissions (U.S. EPA 2011).

Swine Production Demographics

Swine are produced primarily in the Midwest, the Texas/Oklahoma Panhandle, and North Carolina (fig. 10). Since the early 1990s, the size and number of pork producers have changed significantly; swine farms have become fewer in number but larger in size, keeping the national inventory of pigs fairly steady. More than 60% of swine continue to be grown in Corn Belt states. North Carolina is the country's second largest pork-producing state, but its pork production has been limited for 15 years by its moratorium on hog farms that use a lagoon and spray field for manure treatment.

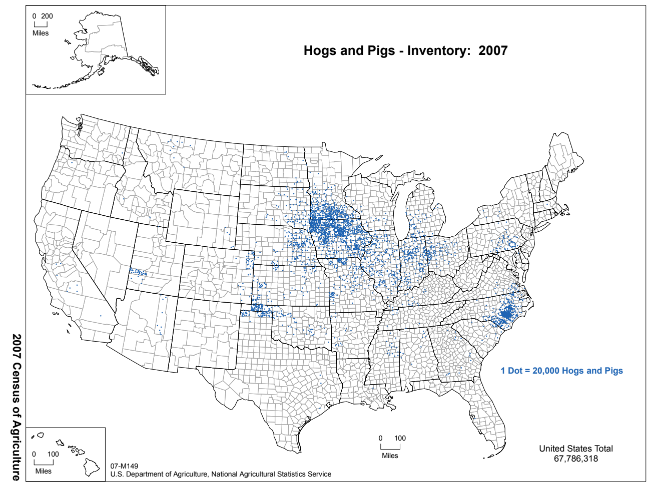

Figure 10. U.S. hog and pig inventory
Source: NASS 2010

Current management systems

Swine production is usually separated into a number of phases, starting with sow breeding and farrowing (giving birth), nursing, and finishing operations, as shown in table 10. Different phases require different management practices.

Table 10. Phases of swine operations

Gestation	Sows experience gestation and lactation cycles throughout most of their life cycle. Sows have a gestation period of 114 days (3 months, 3 weeks, 3 days). This gestation period allows sows to farrow at least twice a year.
Farrowing	Just before giving birth, or farrowing, sows are normally moved into a temperature-controlled "farrowing room." Sows typically farrow 8 to 12 piglets, which are usually placed in a farrowing pen or crate that restricts their movement.
Nursery	After weaning, pigs are usually placed on slotted floors in a temperature-controlled "nursery room." The pigs enter a nursery site at 2 to 3 weeks of age, when they weigh 4 to 5.5 kg, and they exit weighing approximately 23 kg. They typically spend 6 to 10 weeks in a nursery site.
Finishing	In this phase, pigs are fed until they reach market weight of 114 to 125 kg. Some finishing operations are housed indoors, some have curtain-sided walls that allow for natural ventilation, and a small percentage use outside lots.

Diet

Swine diets usually include ground corn to supply energy and soybean meal to provide protein. Rations can be tailored to optimize health and growth at each phase of the pig's life. Phase feeding, shown in table 11, is a common practice.

Table 11. Swine rations

Gestation	A gestation diet is fed to gestating sows as well as to breeding boars. This ration includes vitamins and minerals to meet daily requirements, and usually contains 13% to 16% crude protein (CP).
Prestarter	A pre-starter diet is given to pigs entering the nursery facility and is continued until they have reached 5 to 6 weeks of age. This ration usually includes dried milk products and contains 20% to 22% CP.
Starter	Pigs are self-fed on a starter diet as early as 5 weeks of age. Starter diets have a lower level of lysine than non-starter diets. Most starter rations contain 18% to 21% CP.
Grower	A grower diet is fed to pigs weighing from 23 to 55 kg. This ration usually contains 15% to 16% CP.
Finisher	A series of finisher diets ("step up" diets) are fed to pigs weighing from 55 kg to ~114–125 kg (market weight). This ration generally contains 13% to 14% CP. The finishing pigs are typically on a self-feeder.

Source: Adapted from CEART 2011.

Today, most sows spend their entire lactation period in a farrowing crate and their breeding and gestation in another crate. Outdoor systems are used primarily for gestating sows; outdoor farrowing and finishing is less common. USDA NAHMS (2001) estimated that approximately 19% of U.S. farms keep gestating sows outdoors, whereas outdoor systems are used at only 6% of the sites with farrowing or finishing pigs. Historically, pigs were reared in a pasture, and each sow had its own farrowing hutch. In the 1970s, confinement housing became the norm because of pigs' improved growth rate, comfort, and health and because of environmental compliance standards and economic considerations. In the early 1990s, 82% of swine in the United States were reared in some kind of confinement (Safley et al. 1992). Barker (1996) estimated that open dirt lots or pastures were used to finish approximately 20% to 30% of the market swine and to maintain about 30% to 40% of the breeding stock. Today, social pressure, most often from consumers through retailers, is driving the switch from crate to pen housing for gestating sows.

Pigs are commonly reared on totally or partially slatted floors to allow urine and manure to pass through into a pit or gutter system. The partially slatted floor usually consists of a two-thirds solid concrete and one-third slatted floor. The use of bedding and dry manure management systems are an alternative to the traditional slatted-flooring indoor systems. Hoop-style[2] finishing houses have become increasingly popular. By 2001, Iowa farmers had built approximately 2,100 hoop barns, 90% of them for finishing pigs (Honeyman, Kliebenstein, and Harmon 2001). The bedded hoop barns can also be used for gestating (Brumm et al. 1999) and weaning sows and in finishing systems (Larson, Honeyman, and Harmon 2003).

Until the 1990s, swine production systems were usually located on a single site, and pigs in different phases of production were housed close to one another. Now many swine operations are two- or three-site systems in which pigs are housed in different production phases at different sites to minimize contact and thereby reduce health concerns and improve biosecurity. A two-site system places breeding and gestation operations at one site, and farrowing/nursery and growing/finishing operations at another site; three-site systems also place the nursery at a separate site. Production systems (modes of management) for swine operations include the following:

- **Farrow-to-finish operation**. A production system that contains all production phases at one place, from breeding to gestation to farrowing to nursing to grow-finishing to market. The entire production period takes 10 to 11 months with 4 months for breeding and gestation, plus 6 or 7 months for the litter to reach market weight.
- **Sow operation**. This operation involves farrowing sows and selling the piglets to a nursery operation when they are weaned at about 10 to 12 weeks.
- **Finishing operation**. This operation purchases feeder pigs from a feeder pig operation and feeds them until they attain market weight.

Some operations are nursery operations that rear baby pigs in a temperature-controlled indoor environment where they grow to about 23 kg before being sold or transferred as feeder pigs to a finishing operation. Due to lack of emissions data from nurseries, these facilities are considered to be part of farrow-to-finish operations for the purposes of this document.

In USDA's National Animal Health Monitoring System (NAHMS) study, about 40% of sites had gestation or farrowing phases; more than 80% of sites had a finishing phase (USDA 2008b). Distribution of the three production systems is presented in figure 11.

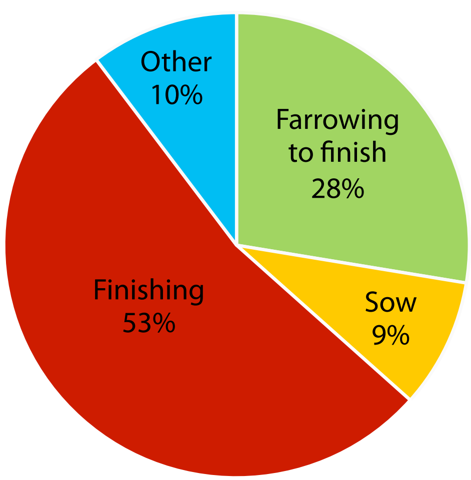

Figure 11. Distribution of the three production systems in the United States
Source: USDA 2008b

2. A hoop barn or house is a plastic roof built over a flexible piping, which requires no center supports and which has a clear span.

Kephart et al. (2001) have estimated the quantities of manure from each of the three production systems at small operations. The manure output is about 725 gal/week for a sow operation with 20 sows and about 1,200 gal/week for a finishing operation with 100 heads. The manure output for a farrowing-to-finish operation with 20 sows and the pigs they farrow is about 2,000 gal/week (Kephart et al. 2001).

The manure management systems associated with swine operations have the basic elements of collection, storage, treatment, transport, and utilization. About 75% of swine facilities in the United States use anaerobic or liquid-slurry systems for manure holding or disposal (Harper, Sharpe, and Parkin 2000). Manure can be collected and stored in an under-floor pit, discharged to a separate storage facility, or flushed to an anaerobic lagoon. The first two methods are prevalent in the Midwest; anaerobic lagoons are common in the Southeast. Common swine manure handling systems are presented in table 12.

Table 12. Common swine manure handling systems

Solid manure handling	Swine manure was historically collected with bedding material used to absorb urine or deposited directly on the ground by grazing pigs or pigs in drylot. The solid manure yields nutrient-rich fertilizers and is normally surface applied or incorporated into soil with a farm tillage operation shortly after spreading. Composting is an option for solid manure management.
Liquid manure storage	Most swine manure is handled as a liquid collected in shallow pits or gutters under slatted floors, which are periodically flushed to outside storage. Another system is to store the manure for up to a year in houses with 4- to 10-foot-deep storage pits. Liquid manure from storage can be surface applied or incorporated into soil during or shortly after application.
Lagoon operation	Lagoons are operated to promote anaerobic digestion of organic material in liquid manure. To function properly, lagoons require dilution water, which is often obtained from the water spillage and misting that normally occur during operation of the housing system. A properly designed and operated treatment lagoon is much larger and more expensive than liquid manure storage with the same storage time, and the organic solids are much less concentrated in the liquid. Lagoon effluent is normally applied to cropland by spray irrigation systems.

Emissions

Manure management emissions

GHG emissions from swine operations mainly include methane (CH_4) and nitrous oxide (N_2O). The leading emissions sources from swine production are

- manure storage and treatment (CH_4 and small amounts of N_2O)
- land application of manure (N_2O)
- enteric fermentation (CH_4)

Methane is produced through anaerobic biochemical decomposition of feed within an animal's digestive system and by the collection, storage, and land application of manure. Monogastric livestock such as swine produce relatively lower CH_4 emissions than ruminant livestock because much less CH_4-producing fermentation takes place in their digestive systems. The IPCC has reported on the fraction of gross energy in feed converted to CH_4 for swine. Its "Tier 1" approach default CH_4 enteric fermentation emissions factor is 1.5 kg yr^{-1} hd^{-1} (36 kg yr^{-1} hd^{-1} in CO_2e) for swine production in developed countries (IPCC 2006). Its Tier 1 approach default manure management CH_4 emissions factors for swine operations are listed in table 13.

Table 13. Manure management CH_4 emissions factors for swine operations in North America

	CH_4 emissions factors by average annual temperature (kg CH_4/hd/yr)		
	Cool (≤14°C)	Temperate (15–25°C)	Warm (≥26°C)
Market swine	10–12	13–20	22–23
Breeding swine	19–23	24–39	41–45

Source: Adapted from table 10.14, IPCC 2006.

The CH_4 emissions from manure management depend on the amount of excreted volatile solid (VS), the maximum CH_4 producing capacity for the manure produced (B_0), and CH_4 conversion factors (MCF) that reflect the percentage of VS actually converted to CH_4. To improve accuracy in estimation of CH_4 emissions from manure management, the IPCC "Tier 2" approach can be used. This approach requires detailed information on animal characteristics and manure management. The manure management CH_4 emissions factor can be estimated using the following equation (IPCC 2006):

$$EF = VS \cdot 365 \cdot B_0 \cdot 0.67 \cdot MCF$$

Where: EF = manure management CH_4 emission factor, kg CH_4/hd/yr; VS = daily excreted volatile solid, kg VS/hd/d; B_0 = maximum CH_4 producing capacity from manure produced, m^3 per kg of VS; MCF = CH_4 conversion factors that reflect the percentage of VS actually converted to CH_4 compared to B_0, %. The factor 0.67 kg m^{-3} is conversion factor of m^3 CH_4 to kg CH_4. The default values of VS, B_0 and MCFs for swine are provided in IPCC (2006).

The spread in estimates of B_0 reported in the literature is significant. This spread is likely to reflect dependence on diet and straw content, which vary significantly from farm to farm but primarily from country to country.

Most of the N_2O emitted by swine production systems originates from microbial decomposition of manure. Once manure has been land applied to soil-crop systems, further complex biochemical processes (e.g., nitrification and denitrification) can produce N_2O from the nitrogen (N) in manure. The emission of N_2O from manure during storage and treatment depends on the N and carbon (C) content of manure and on the storage time and type of treatment. Direct N_2O emissions occur through combined nitrification and denitrification of N contained in the manure. The production and emission of N_2O from managed manures requires the presence of nitrites or nitrates in an anaerobic environment preceded by aerobic conditions necessary for the formation of these oxidized forms of N. Indirect emissions result from volatile N losses that occur primarily in the forms of NH_3 and NO_x. Direct N_2O emissions from manure management are estimated from the N excretion rate and an emission factor. Default emissions factors for direct N_2O emissions from the manure management system are provided in IPCC Tier 1 and Tier 2 approaches. The Tier 2 approach uses country-specific N excretion rates. The IPCC (2006) default emissions factor for direct N_2O emissions from managed soil is 1% (0.01 kg N_2O-N per kg applied N).

The principal factors affecting CH_4 emissions from livestock manure are the amount of manure produced and the portion that decomposes anaerobically. Primary determinants of the extent of anaerobic decomposition are the type of manure management system used and the climate (principally, temperature). The amount of N_2O released depends on the system and duration of waste management. Because N_2O production requires an initial aerobic reaction and then an anaerobic process, researchers theorize that dry and aerobic management systems may provide an environment relatively conducive for N_2O release. Factors influencing the land emissions of N_2O include temperature, precipitation and soil moisture, and application method (Eagle et al. 2012).

Emissions data from the three production systems

1. Farrow-to-finish systems

GHG emissions data from farrow-to-finish operations in North America are summarized in table 14. The average CH_4 emissions rates from the studies included in table 14 are 8.9±5.7 kg CH_4/hd/yr (213±137 kg CO_2e/hd/yr) for swine building and 7.3±6.9 kg CH_4/hd/yr (175±166 kg CO_2e/hd/yr) for manure storage facilities. These numbers are slightly lower than the IPCC (2006) default manure management CH_4 emissions factors in table 13. No data are available for N_2O emissions from swine building; for manure storage facilities, the highest observed N_2O emissions rate was only 6.8 kg CO_2e/hd/yr (Harper et al. 2004).

Table 14. Measured GHG emissions from farrow-to-finish operations in North America

Emissions sources	References	Emissions rates in original units			Emissions rates (kg yr^{-1} hd^{-1})		Emissions rates (kg yr^{-1} hd^{-1} in CO_2e)	Comments
		Original units	CH_4	N_2O	CH_4	N_2O		
Swine building	Sharpe and Harper 2001	g d^{-1} hd^{-1}	6.9±3.4	-	10.6	-	254	Pits beneath slatted floor flushed every 8 h, winter
		g d^{-1} hd^{-1}	29.2±6.7	-	13.6	-	326	Pits beneath slatted floor flushed every 8 h, summer
		g d^{-1} hd^{-1}	37.2±1.4	-	2.5	-	60	Pits beneath slatted floor flushed every 8 h, summer
Manure storage facilities	Lague et al. 2005	g CO_2e d^{-1} kg^{-1} pig	5.8	neg.	4.6	neg.	110	Uncovered concrete tank
		g CO_2e d^{-1} kg^{-1} pig	6.62	0.02	5.3	0.001	127	Uncovered earthen manure basin
		g CO_2e d^{-1} kg^{-1} pig	1.25	neg.	1	neg.	24	Covered earthen manure basin
	Harper, Sharpe, and Parkin 2000	kg d^{-1} ha^{-1}	125.8	neg.	24	neg.	576	Anaerobic lagoon
	Harper et al. 2004	kg N d^{-1} ha^{-1}	-	0.3	-	0.022	6.8	Anaerobic lagoon
	Park et al. 2006	kg yr^{-1} hd^{-1}	6.7	3.6	6.7	0.004	162	Liquid manure storage tank in cold climate
	Sharpe and Harper 1999	kg yr^{-1} hd^{-1}	5.6	-	5.6	-	134	Anaerobic lagoon
	Sharpe, Harper, and Byers 2002	kg yr^{-1} hd^{-1}	6.0	-	6	-	144	Anaerobic lagoon
	Shores et al. 2005	kg d^{-1} $site^{-1}$	122.7	-	5.5	-	132	Manure storage basins

2. Sow systems

The measured GHG emissions data from sow operations in North America are summarized in table 15. The average CH_4 emissions rates in sow operations from the studies included in the table are 30.7±31.3 kg CH_4/hd/yr (737±751 kg CO_2e/hd/yr) for manure storage facilities. These numbers are much higher than those in farrow-to-finish operations. The N_2O emissions from sow systems are negligible in most studies; the exception is Harper et al. (2004), which reported N_2O emissions rates of 25 kg CO_2e/hd/yr from an anaerobic lagoon.

Table 15. Measured GHG emissions from sow operations in North America

Emissions sources	References	Emissions rates in original units			Emissions rates (kg yr^{-1} hd^{-1})		Emissions rates (kg yr^{-1} hd^{-1} in CO_2e)	Comments
		Original units	CH_4	N_2O	CH_4	N_2O		
Swine building	Ball and Mohn 2003	g d^{-1}	21.6±1.4	-	0.66	-	15.8	Non-pregnant sow, barley-based diets
		g d^{-1}	8.8±1.4	-	0.27	-	6.5	Non-pregnant sow, corn-based diets
	Lague et al. 2005	g d^{-1} kg^{-1} pig	0.63	neg.	41.4	neg.	994	Farrowing, liquid manure management
		g d^{-1} kg^{-1} pig	0.27	neg.	14.8	neg.	355	Gestating, liquid manure management
		g d^{-1} kg^{-1} pig	1.96	neg.	12.9	neg.	310	Nursery, liquid manure management
		g d^{-1} kg^{-1} pig	0.1	neg.	6.6	neg.	158	Farrowing, liquid manure management
		g d^{-1} kg^{-1} pig	0.07	neg.	3.8	neg.	91	Gestating, liquid manure management
		g d^{-1} kg^{-1} pig	0.39	neg.	2.6	neg.	62	Nursery, liquid manure management
	Sharpe and Harper 2001	g d^{-1} hd^{-1}	46.2±2.8	-	16.9	-	406	Farrow to wean, pull-plug system beneath slatted floor flushed every 7–8 d, summer
	Zhang et al. 2007	g d^{-1} AU^{-1}	184±170	neg.	24.2	neg.	581	Farrowing, liquid manure stored in under-floor shallow gutters and removed every 3 wks
		g d^{-1} AU^{-1}	351±204	neg.	46.1	neg.	1106	Farrowing, liquid manure stored in under-floor shallow gutters and removed every 3 wks
		g d^{-1} AU^{-1}	118±119	neg.	12.9	neg.	310	Gestating, liquid manure stored in under-floor shallow gutters and removed every wk
		g d^{-1} AU^{-1}	73±51	neg.	8	neg.	192	Gestating, liquid manure stored in under-floor shallow gutters and removed every wk
Manure storage facilities	Harper et al. 2004	kg N d^{-1} ha^{-1}	-	0.4	-	0.081	25	Anaerobic lagoon
	Kaharabata, Schuepp, and Desjardins 1998	kg yr^{-1} m^{-2}	56.5±11.3	-	78.3	-	1879	Sow, hog, and piglet; cereal grain and corn diet; above-surface open manure-slurry tank
	Sharpe, Harper, and Byers 2002	kg yr^{-1} hd^{-1}	1.6	-	1.6	-	38	Anaerobic lagoon
	Zahn et al. 2001	kg d^{-1} $site^{-1}$	831.0	-	16.4	-	394	Farrow to feeder, lagoon systems with anoxic photosynthetic blooms where site refers to the entire farm facility
	Zhang et al. 2007	g d^{-1} m^{-2}	44±27	neg.	45.9	neg.	1102	Farrowing, earthen manure storage
		g d^{-1} m^{-2}	30±25	neg.	11.4	neg.	274	Farrowing, earthen manure storage with negative air pressure covered

Note: AU = animal unit; 1 AU = 454 kg liveweight.

3. Finishing operations

The measured GHG emissions data from finishing operations in North America are summarized in table 16. The average CH_4 emissions rates for swine buildings from the 10 studies included in the table are 3.2±2.8 kg CH_4/hd/yr (77±67 kg CO_2e/hd/yr), which is much lower than the IPCC (2006) default manure management CH_4 emissions factors in table 13 (10 to 23 kg yr^{-1} hd^{-1} for market swine). The average CH_4 emissions rates for manure storage facilities from the 6 studies included in table 16. are 21.6±20.6 CH_4/hd/yr (518±494 kg CO_2e/hd/yr), which is comparable with the IPCC (2006) default manure management CH_4 emissions factors. The highest observed N_2O emissions rate was only 11 kg CO_2e/hd/yr (Lague et al. 2005).

Table 16. Measured GHG emissions from finishing operations in North America

Emissions sources	References	Emissions rates in original units			Emissions rates (kg yr^{-1} hd^{-1})		Emissions rates (kg yr^{-1} hd^{-1} in CO_2e)	Comments
		Original units	CH_4	N_2O	CH_4	N_2O		
Swine building	Ball and Mohn 2003	g d^{-1}	15.0±1.1	-	0.46	-	11	Finishing, barley-based diets
	Desutter and Ham 2005	kg yr^{-1} hd^{-1}	1.5	-	1.5	-	36	Finishing, slatted floor with under-floor pits, drained every 5-7 d
	Kai et al. 2006	g d^{-1} AU^{-1}	14	-	0.5	-	12	Grower pig, slatted floor and recharge pit
	Lague et al. 2005	g d^{-1} kg^{-1} pig	0.14	0.002	2.4	0.034	68	Grower-finisher, liquid manure management
		g d^{-1} kg^{-1} pig	0.24	neg.	4	neg.	96	Grower-finisher, liquid manure management, partially slatted floor
		g d^{-1} kg^{-1} pig	0.43	0.001	7.2	0.017	178	Grower-finisher, liquid manure management, fully slatted floor
	Li, Powers, and Hill 2011	g d^{-1} hd^{-1}	3.2±0.3	0.8±0.2	1.2	0.024	36	Grow-finish, manure cleaned twice weekly, corn control diet
		g d^{-1} hd^{-1}	5.3±0.3	0.8±0.2	1.9	0.024	53	Grow-finish, manure cleaned twice weekly, diet containing 20% DDGS and inorganic trace minerals
		g d^{-1} hd^{-1}	6.2±0.3	0.8±0.2	2.3	0.024	63	Grow-finish, manure cleaned twice weekly, 2 diets containing 20% DDGS with organic trace minerals
	Ni et al. 2008	g $d^{-1}$$AU^{-1}$	36.2±2.0	-	4	-	96	Finishing barn with shallow manure flushing system
		g $d^{-1}$$AU^{-1}$	28.8±1.8	-	3.2	-	77	Finishing barn with shallow manure flushing system
	Pepple et al. 2010	g d^{-1} hd^{-1}	25±1.7	neg.	9.1	neg.	218	Wean to finish, deep pit, non-DDGS
		g d^{-1} hd^{-1}	23.4±1.6	neg.	8.5	neg.	204	Wean to finish, deep pit, DDGS
	Powers, Zamzow, and Kerr 2008	g d^{-1} hd^{-1}	6.2	-	2.3	-	55	Grow-finish, manure cleaned twice weekly
	Zahn et al. 2001	kg d^{-1} $site^{-1}$	52.8	-	1.4	-	34	Feeder to finish, confinement buildings with under-slat storage
	Unpublished study at MSU 2009	g d^{-1} hd^{-1}	1.75±0.01	-	0.6	-	14	Grow-finish, manure cleaned twice weekly
Manure storage facilities	Clark et al. 2005	g d^{-1} m^{-3}	42	neg.	0.2	neg.	5	Liquid swine manure storage vessel
	Desutter and Ham 2005	kg yr^{-1} hd^{-1}	8.3	-	8.3	-	199	Anaerobic lagoon
	Park, Wagner-Riddle, and Gordon 2010	mg s^{-1} m^{-2}	1.75	-	14.1	-	338	Liquid manure storage tank
	Pelletier et al. 2004	g CO_2e d^{-1} kg^{-1} pig	6.4	-	5.1	-	122	Liquid swine manure storage tank
	Shores et al. 2005	kg d^{-1} ha^{-1}	223.9	-	25	-	600	Anaerobic lagoon
		kg d^{-1} ha^{-1}	447.31	-	49.9	-	1198	Anaerobic lagoon
		kg d^{-1} ha^{-1}	209.23	-	23.4	-	562	Anaerobic lagoon
		kg d^{-1} ha^{-1}	118.08	-	13.2	-	317	Anaerobic lagoon
		kg d^{-1} ha^{-1}	580.75	-	64.8	-	1555	Anaerobic lagoon
	Zahn et al. 2001	kg d^{-1} $site^{-1}$	466.1	-	12	-	288	Lagoon systems without anoxic photosynthetic blooms

Summary

Manure management CH_4 emissions appear to represent the most significant part of the total CH_4 and N_2O emissions from swine operations, followed by N_2O emissions from manure land application. In IPCC (2006), the manure management CH_4 emissions factors for market swine operations in North America are 10 to 23 kg CH_4/hd/yr (240 to 552 kg CO_2e/hd/yr) (see table 13.). The default CH_4 enteric fermentation emissions factor for swine production in developed countries is only 1.5 kg CH_4/hd/yr (36 kg CO_2e/hd/yr), which is less than 15% of the above manure management CH_4 emissions factors. For the most common swine manure system (pit storage below animal confinement), the IPCC (2006) default emissions factor for direct N_2O emissions from the manure management system is 0.2% (0.002 kg N_2O-N per kg excreted N), which indicates N_2O emissions rates of 0.02 to 0.05 kg N_2O/hd/yr (6 to 16 kg CO_2e/hd/yr). The N_2O emissions from the manure management system in CO_2e unit are less than 7% of the above manure management CH_4 emissions factors. The IPCC (2006) default emissions factor for direct N_2O emissions from managed soil is 1% (0.01kgN_2O-N per kg applied N). Assuming all excreted manure is to be land applied, the N_2O emissions rates from land application would be 0.1 to 0.25 kg N_2O/hd/yr (31 to 78 kg CO_2e/hd/yr), which is less than 33% of the above manure management CH_4 emissions factors in CO_2e units.

The many studies of manure management CH_4 emissions rates from swine operations reveal wide variations in those rates among different production systems, manure handling practices, and climate conditions. Results of a meta-analysis show that farrowing operations generate the highest CH_4 emissions, and finishing operations generate the lowest CH_4 emissions as compared with other stages of swine production (Liu, Powers, and Liu 2011). Those results also show that CH_4 emissions from swine buildings in sow systems examined in various studies can vary 170-fold, from 6.5 to 1,106 kg yr^{-1} hd^{-1} in CO_2e (table 16). This large variation may indicate large opportunities for mitigation or that studies are not comparable because of undisclosed factors.

Opportunities for Mitigation

Feeding strategies

Control of GHG emissions is directly related to nutrient efficiency in livestock (Bhatti et al. 2005). Feeding strategies that mitigate GHG emissions in the swine industry mainly include those that enhance feed utilization efficiency, reduce nutrient excretions, and shift nutrient excretions from urine to feces.

Low-protein diets

Blending nutrient ingredients in feed to supply each nutrient ingredient at exactly the level required by an animal is difficult. Therefore, feed usually supplies far more protein than is needed to satisfy the requirement for the most limiting nutrient. Matching dietary nutrients with the requirements of the pig reduces the excretion of excess nutrients, such as nitrogen and carbon. Lower nutrient availability, in turn, could reduce GHG emissions from manure.

Low-protein diets can be used when supplemental amino acids are formulated to provide the limiting amino acids in the diet. Several studies reported reductions in CH_4 and CO_2 emissions resulting from low-protein diets (Velthof et al. 2005; Atakora, Moehn, and Ball 2004; Moehn, Atakora, and Ball 2004; Atakora, Moehn, and Ball 2003; Lague 2003; Ball and Moehn 2003; Misselbrook et al. 1998). Moehn, Atakora, and Ball (2004) claimed that for every 10% reduction in dietary CP, a 10% reduction in CO_2e emissions from pigs resulted.

The emission of CH_4 was significantly related to the content of dry matter, total C, and volatile fatty acids (VFA) in the manure. Misselbrook et al. (1998) suggested that 50% of the reduction in CH_4 emissions from the slurry observed when pigs were fed the lower CP diet was probably the result of the reduced VFA content of the slurry. Decreasing the CP content has the largest potential to simultaneously decrease NH_3 and CH_4 emissions during manure storage and N_2O emissions from soil (Velthof et al. 2005).

Every one percentage unit of dietary CP level reduction can result in a 6%–9% reduction in N excretion (Sutton et al. 1999; Kendall et al. 1998; Hobbs et al. 1996; Kerr 1995; Aarnink, Hoeksma, and van Ouwerkerk 1993). Several studies reported that the reduction of total N excretion through low protein diets was mainly through the reduction in urinary N excretion, driven by increased dietary amino acids plus corn substitution for soybean meal and other feed N sources (Canh et al. 1998; Gatel and Grosjean 1992). Urinary nutrients are mostly inorganic in nature and are easily volatile, whereas fecal nutrients are mostly organic in nature. Biological conversion of nutrients from stable organic form to inorganic form is a slow process (Vaddella et al. 2010). Several studies also reported reduced manure pH with low-protein diets (Portejoie et al. 2004; Kendall et al. 1998; Canh et al. 1998). Reduced N excretion is expected to have the potential to reduce N_2O emissions from manure. However, this effect has not been studied by direct measurements of N_2O emissions from manure.

Feed ingredients

Fiber

Jensen and Jørgensen (1994) found that high-fiber diets increased CH_4 emissions. Jørgensen (2007) claimed that the production of CH_4 depended on fiber origin; however, this effect varies widely among animals. Ball and Moehn (2003) found that CH_4 production was greater for barley-based diets than for corn-based diets. Growing pigs fed diets varying in total fiber content (2.8%–40%) had a CH_4 production equivalent to 0.1%–1.3% of digested energy.

Distillation byproducts

Powers et al. (2008) observed that inclusion of distillers dried grains with solubles (DDGS) in a corn diet lowered CH_4 emissions from swine housing. However, Li, Powers, and Hill (2011) reported that feeding 20% DDGS to grow/finish pigs increased CH_4 emissions. Pepple et al. (2010) observed that DDGS has no significant effect on GHG from manure storage.

Fermentable carbohydrates

Aarnink and Verstegen (2007) found a close relationship between fermentable carbohydrates in the diet and CH_4 production. Increasing the fermentable carbohydrate level in the diet to lower the pH of feces and manure, and consequently NH_3 emissions, will, at the same time, increase CH_4 production (Aarnink and Verstegen 2007).

Oil additives

Christensen and Thorbek (1987) reported that flatus production may be reduced not only by changing the composition of dietary carbohydrates, but also by including polyunsaturated oil in the diet of simple-stomached animals and humans. In a feedlot study, Mathison et al. (1997) reported 33% reductions in CH_4 emissions when a diet containing 85% concentrate was added with 4% canola oil.

Manure management systems

Because CH_4 is a byproduct of anaerobic microbiological decomposition processes, implementing manure collection practices that prevent or limit such processes can reduce GHG emissions. Because of its poor solubility in water, CH_4 emits from manure as soon as CH_4 is produced (Monteny, Bannink, and Chadwick 2006). Furthermore, the degree of anaerobic bacteria fermentation and therefore the amount of CH_4 emissions depend on pH value, slurry temperature, retention time, and the presence of inhibiting compounds (Zeeman 1991; Huther, Schuchardt, and Willke 1997). Ni et al. (2008) claimed that the design of swine barns and the management of stored manure influence CH_4 emissions. Zahn et al. (2001) indicated that the manure management environment, and specifically loading rate, may significantly influence the flux rate of CH_4 from stored swine manure.

Time in pit

Long-term manure storage in deep-pit pig barns may result in increased gas emissions. Sharpe and Harper (2001) claimed that the longer the retention time, the greater the decomposition of manure before it is pumped into the lagoon. Kai, Kaspers, and van Kempen (2006) suggested that complete removal of manure from the pig house, e.g., by flushing, helps lower CH_4 emissions from pig housing. Frequent removal of manure from indoor storage pits—through quick land application or sufficient outdoor storage facilities—will reduce GHG emissions (Osada et al. 1998).

Temperature

Many researchers have identified temperature as an important factor for CH_4 emissions from manure storage facilities (Amon et al. 2007; Haeussermann et al. 2006; Desutter and Ham 2005; Moller, Sommer, and Ahring 2004; Sharpe and Harper 1999; Husted 1994; Cullimore, Maule, and Mansuy 1985). Low temperatures can suppress microbial activities and metabolism and therefore production of CH_4. Cullimore, Maule, and N. Mansuy (1985) observed that CH_4 production from pig manure linearly increased with increasing effluent temperature when $T < 26°C$, and Husted (1994) observed that CH_4 emissions from solid pig manure peaked at 35 to 45°C. Massé et al. (2003) claimed that the effect of temperature on CH_4 production was more important for swine manure than for dairy cow manure.

Temperature can also affect gas emissions from swine buildings through its influence on ventilation. High temperatures induce high ventilation rates (Dong et al. 2009; Ni et al. 2008; Zhang et al. 2007). Blanes-Vidal et al. (2008) reported that the correlation between averaged ventilation flow and CH_4 emissions on an hourly basis was positive with an $R^2 = 0.79$. Ni et al. (2008) claimed that temperature is an important factor for both CH_4 and CO_2 emissions from swine buildings.

Slurry cooling as well as lowering indoor temperatures and air exchange rates are proactive methods to reduce CH_4 emissions (Monteny, Groenestein, and Hilhorst 2001). Sommer, Petersen, and Møller (2004) reported that pig slurry cooled by 10°C reduced CH_4 emissions by 21% when compared with uncooled slurry. Bates (2001) reported that CH_4 emissions reductions of 60%–100% could be achieved with a drop of manure temperature from 20°C to 10°C. On the other hand, Guarino, Costa, and Porro (2008) observed that reducing room temperature negatively affected N_2O emissions from swine buildings.

Floor openness

Lague (2003) reported that the CH_4 production rate is higher with the fully slatted floor room than with the partially slatted floor room. The larger area of contact between the manure and the air likely increases CH_4 emissions (Lague 2003). Steed and Hashimoto (1994) observed contrary results; they found higher methane conversion factors in closed systems than in systems open to the atmosphere, a phenomenon they attributed to inhibition of methanogenesis by oxygen.

Solid-liquid separation

In-barn solids/liquids separation systems and separate handling and storage of the two product streams have been studied as ways to mitigate gas emissions. Su, Liu, and Chang (2003) reported that separating liquids and solids immediately after pig houses are washed (to prevent manure from becoming slurry) can reduce biogas production by 62% and also prevent CH_4 production. However, Dinuccio, Balsari, and Berg (2008) showed that solid-liquid separation of pig slurry reduced NH_3 losses but increased CH_4 emissions by 3% and CO_2 emissions by 10% compared with the storage of rough slurry.

Bedding systems and solid manure storage

Cabaraux et al. (2009) and Dourmad et al. (2009) reported that CH_4 and N_2O emissions decreased in a sawdust bedding system as compared with a fully slatted floor/pit system. Philippe et al. (2007) reported that straw bedding systems produce more NH_3 and N_2O than slatted floors but have no effect on CH_4 emissions. The effect of bedding material on CH_4 emissions has also been investigated. Nicks et al. (2003, 2004) reported that pig houses with saw dust-based litter emitted 33% less CH_4 than straw-based litter systems.

Monteny, Bannink, and Chadwick (2006) indicated that solid manure storage would mitigate N_2O emissions with compaction of solid manure. However, they also pointed out that the anaerobic conditions due to compaction could increase CH_4 emissions. In contrast, Maycher (2003) stated that higher oxygen levels exist in solid manure handling, which results in greater CO_2 emissions offsetting CH_4 emissions.

Covered manure storage

The use of covers has been proposed to reduce CH_4 and CO_2 emissions from manure storage facilities. Lague et al. (2003) observed that the presence of a blown chopped straw cover on these facilities reduced GHG emissions. Amon et al. (2007) also demonstrated that a solid cover was effective in mitigating GHG and NH_3 emissions from stored pig slurry. Safley and Westerman (1988) found that CH_4 emissions rates may vary depending on the areas of lagoon covered. Hansen, Henriksen, and Sommer (2006) reported that covers reduced CH_4 emissions by 88%. However, it was observed that straw covers on manure storage structures can increase CH_4 emissions, and straw or swelled-clay covers on manure storage structures can increase N_2O emissions (Lague et al. 2005). Zhang et al. (2007) reported that a negative air pressure cover resulted in no significant reduction in CH_4 emissions in comparison with open earthen manure storage. Guarino et al. (2006) evaluated the effectiveness of five simple floating covers for reducing emissions from pig and cattle slurry. They found that, at the greatest thickness, all the tested covers increased the efficiency of NH_3 and CO_2 emissions reduction, but they found no statistically significant CH_4 emissions reduction from pig slurry (Guarino et al. 2006).

The CH_4 emissions from lagoons are related to wind speed (Sharpe, Harper, and Byers 2002; Sharpe and Harper 1999). Husted (1994) observed a strong dependence of CH_4 emissions rates on air flow through the chamber for pig solid manure but not for pig slurry.

Composting

The composting process may increase emission of N_2O, but some evidence suggests that the subsequent production and emission of N_2O from land-applied composted manure are much lower than those for raw manure. Several studies have concluded that manure composting has the potential to reduce GHGs (Boldrin et al. 2009; Brown, Kruger, and Subler 2008; Zeman, Depken, and Rich 2002). Fukumoto et al. (2003) demonstrated that changing the scale of the compost pile could change CH_4 and N_2O emissions rates during swine manure composting. Another trend is the separation of swine manure solids from liquids for composting. These solids yield fertilizers that may be sold off-site as another source of income for the farm, however the practice is rare because it often becomes a cost center.

Anaerobic digesters

Research on the anaerobic digestion of swine manure is significant. Generating biogas (60%–65% CH_4) for on-farm power or heat production with an anaerobic digester with manure slurry is very effective (Monetny, Bannink, and Chadwick 2006). Biogas generation offsets fossil fuel use in addition to GHG emission reductions.

Other

CH_4 production from slurry can be reduced by adding inhibiting compounds and acids (Berg and Pazsiczki 2003; Amon et al. 2004). Berg and Pazsiczki (2003) demonstrated that 80% reductions in CH_4 emissions could be possible with the addition of lactic acid and limestone. Pelletier et al. (2004) reduced GHG emissions successfully from liquid swine manure using an aerobic-anoxic manure treatment system and a biofilter manure treatment system. Vanotti, Szogi, and Vives (2008) reported that replacing an anaerobic lagoon technology with a clean aerobic technology reduced CH_4 emissions 96.9%. Costa and Guarino (2008) and Guarino et al. (2008) demonstrated that photocatalytic treatment with TiO_2 coating and UA-A light can reduce CH_4 emissions from swine buildings.

Field application of manure

Improved land application practices

Swine manure provides important nutrients to support crop production. However, over-application of swine manure can result in excessive soil N content, increasing N_2O emissions. Timing application to match the crop soil N requirement can mitigate these emissions (Maycher 2003). Frequent application of swine wastes might also mitigate GHG emissions, though the practice is not very practical in areas where crops are not grown year-round (Maycher 2003).

Other land application practices might mitigate GHG emissions from manure land application. These practices include direct injection (liquid manure) or rapid incorporation (by plowing or similar techniques) into soil; proper application depth; good management of water irrigation, land drainage, and tillage practices; and choice of fertilizer form (Monetny, Bannink, and Chadwick 2006; Maycher 2003).

Nitrification inhibitors

Adding certain chemical compounds to fertilizer can inhibit conversion of NH_3 to N_2O (Maycher 2003). This process would increase crop uptake and reduce N_2O emissions. Good nitrification inhibitors include 3, 4-dimethylpyrazole phosphate (DMPP), dicyandiamide (DCD), and Nitrapyrin (Pain, Misselbrook, and Rees 1994). Clark, de Klein, and Newton (2001) reported that 50% N_2O emissions reductions could be achieved with the addition of nitrification inhibitors in fertilizers. In a field study, Dittert et al. (2001) observed that N_2O emissions were reduced by 32% with the addition of DMPP to manure slurry injection. The synthesis of research on mitigation practices by Eagle et al. (2012) found that nitrification inhibitors used in croplands and grasslands resulted in emissions reductions ranging from 0 to 1 tonne (t) CO_2e/ha/yr and a mean of 0.4 t CO_2e/ha/yr.

Mitigation Options with the Greatest Likelihood of Impact on Emissions from Swine Production

Methane emissions from manure management represent the most significant part (more than 65% in CO_2e) of the total CH_4 and N_2O emissions from swine operations. Therefore, the options with the greatest likelihood for success will focus on mitigating these emissions, while ensuring that other GHG emissions will not increase (table 17). Both feeding strategies and practices in manure management systems can help mitigate manure management CH_4 emissions. Switching to optimized bedding systems with dry manure management, composting, covering manure storage, and using anaerobic digesters could all be effective mitigation strategies. Increasing manure removal frequency has shown consistent effectiveness in different studies. Given cost considerations, economic or regulatory incentives may be required to promote wide adoption of some of these strategies. Dietary strategies pose fewer cost concerns, but more research is needed to confirm their consistent effectiveness and their influence on animal performance.

Reducing N_2O emissions from land application of manure represents another critical control point for mitigating farm-level carbon emissions. Improving the application of manure and any additional fertilizers so that rate and timing of application match crop soil N requirements, as well as the use of nitrification inhibitors can improve nitrogen use efficiency, limit N_2O emissions and other nitrogen losses. Concerns related to the use of nitrification inhibitors include cost and the period of time (3 to 4 weeks) for which their application remains viable.

Table 17. Summary of swine mitigation practices for swine GHG emissions, based on the opinions of expert authors

	Mitigation potential	Amount of research	Expert confidence	Potential expense	Action category*
Feeding strategies					
Lower protein diet	10% reduction in protein ~ 10% reduction in CH_4; potential to lower N_2O not studied	Little	Low	Low	Research priority
Feed ingredients	Unclear	Little	Low	Low	Uncertain
Oil additives	Unclear	Little	Low	Low	Uncertain
Manure management					
Reduced time in pit	30% reduction in methane	Moderate		Uncertain	Low priority
Floor openness	Unclear	Little	Low	High	Low priority
Solid-liquid separation	Unclear	Little	Low	High	Research priority
Solid manure storage	Unclear	Little	Low	significant	Uncertain
Bedding materials	N_2O uncertain; CH_4 30%	Little	Low	uncertain	Research priority
Lower temperature and better ventilation	10–20°C drop ~ 20%–60% for CH_4; unclear for N_2O; increase CO_2 from energy need	Moderate	Moderate	High	Uncertain
Covering manure storage	Can be significant depending on cover type, ranges from 88% reduction to an increase	Moderate but all with different cover types	Moderate	Moderate	Research priority
Composting	Significant but variable	Moderate	Moderate	Potential profit	Research priority
Anaerobic digestion	Significant	High	High	High	Ready; high priority

*Action category indicates whether the action (1) is *ready* for integration into programs and protocols as a mitigation practice and is of *high* or *moderate priority*, given its potential; (2) is likely to have significant mitigation potential but is supported by little research, making it a *research priority*; (3) appears to have low mitigation potential or significant implementation barriers and thus is a *low priority* for research or action; or (4) is supported by too little research to make a recommendation and is therefore *uncertain*.

4. MEASUREMENT AND PREDICTION OF GREENHOUSE GAS EMISSIONS FROM LIVESTOCK SYSTEMS

Direct Methane Measurement Options

The major greenhouse gas emitted by livestock systems is methane from enteric fermentation and manure storage. Methods to measure CH_4 emissions from ruminants range from whole-animal respiration calorimetry chambers with various types of gas analyzers to tracer techniques and whole-barn mass-balance techniques (Young, Kerrigan, and Christopherson 1975; McLean and Tobin 1987; Johnson and Johnson 1995). These methods have been summarized (Johnson and Johnson 1995; Kebreab et al. 2006b; Harper, Denmead, and Flesch 2011) and can be broadly categorized as chamber, tracer, and micrometeorological. The method best suited for a given situation depends on the information to be established and on the animals' housing environment and production system.

Chamber methods

Chamber methods are considered to be the gold standard (McLean and Tobin 1987; Johnson and Johnson 1995) in measuring CH_4 (and CO_2) from animals. These methods range from sealed stainless steel chambers to more portable, flexible tunnel-like enclosures. All of these methods involve the creation of a unidirectional air flow that passes over one or more animals from an incoming to an outgoing opening. Outside air is circulated around the animal's head, mouth, and nose, and expired air is collected or sampled. Gaseous exchange is determined by measuring the total airflow through the system and the difference in concentration between inspired and expired air. Gaseous composition of the ingoing and outgoing air from the respiration chamber can be measured using methods such as dual-channel infrared and paramagnetic analyzers (Cammell et al. 1986). Other chambers of similar design and function measure the rate of air flow and the concentration of various gases at the incoming and outgoing vents with direct measure analyzers, including flame ionization and photo-acoustic technology (Sun et al. 2008; Hamilton et al. 2010; Stackhouse et al. 2011). Typically, chambers are constructed of steel and have an air conditioning system to maintain a temperature range of 18 ± 2°C and relative humidity of 60 ± 10%. Although used extensively, whole-animal open-circuit indirect respiration chamber systems are expensive to construct and maintain. Recently, some of these systems have been adapted to use polycarbonate, which is a transparent material and cheaper to construct.

Hood Chamber

Portable chamber-based methods focus solely on eructated gaseous emissions (those coming from the mouth and nostrils of the animal) and heat exchange (Kelly et al. 1994; Odongo et al. 2008). These methods allow the animal to eat, drink, and lay down while secured to a head chamber (a detailed description of which is provided by Odongo et al. 2008). Therefore, these methods are useful to determine the effects of various treatment options on eructated gaseous emissions. Their main disadvantages are their high labor costs, their inability to measure hindgut CH_4, and their unsuitability for use in the pasture. A face mask that measures CH_4 from eructated gases has been developed, but it does not allow the animal to eat, drink, or behave normally (Liang, Terada, and Hamaguchi 1989).

Polythene tunnel

Chambers made of steel or polycarbonate and portable chambers measure one animal at a time in confinement. Therefore, they are not suitable for pasture-based studies. Using similar principles, Lockyer and Jarvis (1995) developed a system constructed from a large polythene tunnel with two small wind tunnels to blow air into and draw air from a larger tunnel. The concentration of CH_4 in the air entering and leaving the tunnel is measured using a gas chromatograph fitted with a flame ionization detector (a detailed description of which is provided by Lockyer and Jarvis 1995). Using the same system, Murray et al. (1999) reported that a peak CH_4 concentration rise to about 10 mL L^{-1} in the tunnel can be detected with an accuracy of about 0.4% over the measurement range. Evaluation of the system showed that it recovers less CH_4 than traditional chambers (Murray et al. 1999). The advantages of the system are that it allows free movement of animals inside the tunnel and is inexpensive to build. However, control of temperature inside the tunnel when ambient temperatures are high is challenging. Due to space limitations, most experiments using this method have been limited to sheep.

Gas tracer method

The most commonly used inert gas tracer is sulphur hexafluoride (SF_6). The rate of SF_6 emission is assumed to be exactly the same as that of CH_4 emission. The method, as described by Johnson et al. (1994a), involves placing a permeation tube containing SF_6 in the rumen, collecting samples from the animal's nose and mouth, and determining CH_4 and SF_6 concentrations by gas chromatography. Methane production is then calculated as the ratio of CH_4 and SF_6 concentrations multiplied by the release rate of SF_6 from the permeation tube.

A tracer method using ethane (C_2H_4) to measure rumen gas kinetics in grazing dairy cows is described by Moate et al. (1997). They reported that C_2H_4 had no effect on the rumen fermentation pattern and was not metabolized. They continuously injected C_2H_4 into the rumen and simultaneously collected rumen gas, which was analyzed for C_2H_4 and principal rumen gases such as CH_4, H_2, CO_2, H_2S, and O_2 to study gas kinetics in rumen headspace. Total CH_4 production can be calculated by dividing the proportion of CH_4 by the proportion of C_2H_4 in the collected gas and multiplying the fraction by the total C_2H_4 infused into the rumen (Mbanzamihigo et al. 2002).

The tracer method is particularly useful for free ranging cattle because it allows CH_4 emissions to be estimated as the animal is grazing at pasture, which in the case of beef cattle could be 5 to 12 months per year (McCaughey, Wittenberg, and Corrigan 1997). Another advantage is that minimal training is required to adapt animals to the use of the apparatus. Limitations include the fact that whole animal emissions are not measured, variability in animal-to-animal measurements is high (Pinares-Patiño et al. 2011), the withdrawal time of animals on release of gas is long, milk produced may need to be discarded, and training in handling tracer gases is required. Furthermore, for a tracer method such as the SF_6 method to work, large upwind sources of CH_4 or SF_6 emissions must be avoided, wind direction must be monitored, and sampling cans must be far enough downwind to allow mixing of CH_4 and SF_6 to allow calculation of CH_4 emissions estimates (Johnson et al. 1994b). When calibrating the rate of release of SF_6, the difference in the mass of SF_6 and that of CH_4 should be considered.

Micrometeorological methods

Micrometeorological methods focus on measuring the flux of gas in the atmosphere relative to the fluxes of animal emissions. These methods use approaches such as mass balance, vertical flux, and inverse dispersion (Harper et al. 1999). The mass balance approach is most suitable for a small area with a well-defined volume, comparable to an outdoor chamber. Given wind speed, ambient gas, and output gas composition monitoring, the emissions rate can be calculated by subtracting the output from input fluxes (Harper, Denmead, and Flesch 2011). Vertical flux methods develop emission rates on the basis of mean vertical atmospheric concentration of gases. These methods can use vertical wind speed and gas concentrations to calculate emission rates. They are most useful when the enclosure above which measurements are taken is homogenous, such as a cattle feedlot that is uniform in construction (Harper, Denmead, and Flesch 2011). Inverse dispersion analysis relates the theoretical relationship between the emissions rates coming from animal enclosures with downwind concentrations of atmospheric gases. In conjunction with upwind concentrations of atmospheric gases representing ambient conditions, an emissions rate can be calculated from animal housing (Harper, Denmead, and Flesch 2011). Judd et al. (1999) also used a non-disturbing micrometeorological flux-gradient technique in which sensible heat was used as a tracer of turbulent transfer. The major advantages of micrometeorological methods are that animals are undisturbed and exhibit normal behavior and that the data provide a snapshot of an entire operation's GHG emissions. The methods' disadvantages include equipment expense and an inability to distinguish emissions from other close emissions sources.

New techniques under development

Archaeol in ruminant feces

A lipid biomarker for archaea, archaeol, is found in feces of ruminants and may potentially be used to calculate CH_4 emissions. Archaeol could be associated with rumen methanogen activity and may indirectly represent methanogenesis associated with enteric fermentation (Gill et al. 2010, 2011). Gill et al. (2011) compared the archaeol concentration in feces of steers fed high forage with that of steers fed high concentrate diets and then compared these concentrations with CH_4 emissions calculated by the SF_6 tracer method. There was some correlation between archaeol concentration and CH_4 production (r = 0.55); furthermore, the fecal archaeol concentration from steers fed the high concentrate diet was significantly lower than that of those fed the high forage diet. The result agrees with what is expected; therefore, measuring fecal archaeol could potentially provide an indirect estimate of CH_4 emissions. The method should be evaluated by using the chamber technique over several days to investigate the correlation of archaeol concentration with CH_4 emissions more precisely. More research on this promising method for predicting CH_4 emissions is needed.

Mobile carbon dioxide and methane analyzers

Madsen et al. (2010) proposed to use CO_2 as an internal marker. The method is based on the assumption that CO_2 production can be estimated from heat production, which in turn can be calculated from the difference between intake of metabolizable energy and energy in products, i.e. weight gain and milk production. This knowledge, combined with simultaneous measurements of the concentration of CH_4 and CO_2, is the basis for the method of quantifying CH_4 production from individual animals. Although it needs to be evaluated using the chamber technique, this method is potentially easy, mobile, and cheap. On the other hand, adapting it to use on pasture-based animals is difficult and, if not automated, it will provide only snapshots of emissions.

An Internet-interfaced system that measures CH_4, CO_2, and other metabolic emissions from individual animals and that tracks changes in emissions over time has been developed by Zimmerman et al. (2011). The system is based on a headstall unit to restrict and control atmospheric mixing. It assigns a radio-frequency identifier to each animal and provides a feed or water dispenser so that the animal voluntarily keeps its head in the correct position to obtain quantitative, representative metabolic gas measurements. A fan unit captures expired and eructated gases. Using a tracer gas, sensors measure CH_4, CO_2, water vapor, molecular hydrogen, hydrogen sulfide, and air flow rate, which are captured in a data acquisition and control system; a remote data link transmits the data to a specified location. The system is undergoing field testing.

Comparison of Measurement Methods

Because the chamber method is considered to be the gold standard, other methods are evaluated against it.

SF_6. In a study by Grainger et al. (2007), the SF_6 apparatus was constructed to simultaneously collect CH_4 emissions from the mouth and rectum, simulating whole animal emissions. Measurements with the SF_6 apparatus were more variable across days and across cows compared with chamber method measurements. However, these differences were small and, overall, CH_4 emission measurements from both techniques were similar. Grainger et al. (2007) therefore concluded that the SF_6 method is reasonably accurate. Boadi, Wittenberg, and Kennedy (2002) and Pinares-Patiño et al. (2011) also compared the SF_6 technique with typical respiration chamber measurements. They found that the SF_6 technique provides measurements that have higher across- and within-animal variation. Hence, this method may require many animals and replications to produce acceptable results. Johnson et al. (1994b) compared CH_4 production by cattle in chambers with values obtained using the SF_6 technique. Although they found CH_4 estimates from the SF_6 technique to be numerically lower (by 7%), the difference was not significant. Most of the techniques use specialized equipment for determination of the concentration of CH_4 and other gases. Therefore, the differences in CH_4 estimates are likely due to sample concentrations rather than to the inaccuracy of measuring equipment.

Polythene tunnel. Murray et al. (1999) compared CH_4 emissions from sheep in a polythene tunnel system and sheep in a respiration chamber. They found that CH_4 production measured in the respiration chamber was 12.9% greater than CH_4 production measured in the polythene tunnel system (in L/kg dry matter intake). Given that recovery of added CH_4 was above 95% for both systems and that no systematic differences in measurement errors were detected, the authors suggested that differences in CH_4 emissions were due to effects of housing during sample collection.

The literature suggests that measurements of emissions from unconfined animals using various techniques agree with an acceptable level of variation. However, the chamber technique, although reliable, might influence emissions rates by changing animals' behavior. Standardization of techniques is required to build a database of emissions from livestock production, which will allow accurate estimation of CH_4 emissions. Data from studies of enteric CH_4 emissions from cattle on various diets need to be collated into this database, which could then be used as a platform to develop prediction models and to identify gaps in knowledge. In addition, measurements taken at different spatial scales need to be standardized, especially when they are used to downscale or upscale emission values.

Modeling Greenhouse Gas Emissions from Livestock

Measurement of CH_4 production in animals requires complex and often expensive equipment; therefore, prediction equations are widely used to calculate CH_4 emissions. Predictions of CH_4 emissions are made using methods that range from simple fixed coefficients (e.g., IPCC 2006; Tier 1) and empirical models (e.g., Mills et al. 2003) to highly complex models (e.g., Dijkstra et al. 1992; COWPOLL). Some models have been developed specifically to predict CH_4 emissions from animals; others have been modified or adapted to calculate CH_4 emissions from rumen fermentation. Models can be broadly classified as statistical or empirical models that relate nutrient intake to CH_4 output directly or as dynamic mechanistic models that attempt to simulate CH_4 emissions on the basis of a mathematical description of ruminal fermentation biochemistry.

Statistical (empirical) models

Empirical models of CH_4 production aim principally to describe the response of the animal to a change in conditions (such as a change in diet). Table 18 summarizes statistical and empirical equations from the literature. Usually, an empirical model is simpler and is more easily and quickly constructed than a mechanistic model. Therefore, depending on objectives, statistical models may be better and more practical. The main disadvantage of an empirical model is that its parameters are usually not biologically meaningful. For example, if the phenomenon of interest was not part of the model's underlying data, predictions may be incorrect.

Table 18. Empirical methane prediction equations (organized by date)

Eq.	Prediction equation for methane (g/d)	Reference
1	75.42 + 94.28 × DMI	Kriss (1930)
2	[−2.07 + 2.63 DMI − 0.105 DMI^2]/0.05565	Axelsson (1949)
3	[1.3 + 0.112 × (EDm) + MN × (2.37 − 0.050 ×(EDm))] / 100 × GEI /0.05565	Blaxter and Clapperton (1965)
4	[2.81 + 0.042 DOMI]/1.4	Murray, Bryant, and Leng (1976)
5	[3.41 + 0.511 × NSC + 1.74 × HC + 2.65 × CEL]/0.05565	Moe and Tyrrell (1979)
6	Eq. 1: 63 + 79 × CF + 10 × NFE + 26 × CP -212 × EE	Kirchgeßner, Windisch, and Müller (1995)[a]
7	Eq. 2: 10 + 4.9 × Milk Yield (kg/d) + 1.5 × $BW^{0.75}$	
8	DEI (0.094 + 0.028 S_{ADFI}/T_{ADFI}) − 2.453 (MN − 1).	Yan et al. (2000)
9	[50 + 0.01 × Milk Yield (kg/d) × 365]/365 × 1000	Corré (2002)
10	Linear 1: [5.93 + 0.92 × DMI]/0.05565	Mills et al. (2003)
11	Linear 2: [8.25 + 0.07 × MEI]/0.05565	
12	Linear 4: [1.06 + 10.27 × forage proportion + 0.87 × DMI]/0.05565	
13	Non-lin. 1: [56.27 − (56.27 + 0) × $e^{(-0.028 \times DMI)}$]/0.05565	
14	Non-lin. 2: [45.89 − (45.89+ 0) × $e^{(-0.003 \times MEI)}$]/0.05565	
15	Non-lin. general $a - (a + b)\, e^{-[(0.0011 \times Starch (kg/d)/ADF] + 0.0045)]ME}$	
16	[45.0 − 0.018 × DMI (g/kg BW/d)2 − 1.84 × C18:2 (% DM) − 84.2 × C≥20 (% DM)] × DMI (kg/d) × 0.6802	Giger-Reverdin, Morand-Fehr, and Tran (2003)
17	20 × Concentrate Intake (kg as fed/d) + 22 × Corn Silage Intake (kg DM/d) + 27 × Grass Intake (kg DM/d)	Schils et al. (2006)
18	Tier 1: 323 for North America, 274 for Europe	IPCC (2006)
19	Tier 2: [0.065 × GEI]/0.05565	
20	Beef: CH_4 (MJ/d) = [2.94 + 0.059 × MEI + 1.44 × ADF − 4.16× lignin (kg/d)]/0.05565	Ellis et al. (2007)[b]
21	Dairy 1: [8.56 + 0.14 × forage (%)]/0.05565	
22	Dairy 2: [3.23 + 0.81 × DMI]/0.05565	
23	Combined beef and dairy: [3.27 + 0.74 × DMI]/0.05565	
24	[(9.75 − 0.05 × Digestibility Rate (%))× GEI]/0.05565	FAO (2010)

Abbreviations are as follows: *a* = theoretical maximum CH_4 output, *b* = minimum CH_4 output ADF= acid detergent fiber (kg/d), BW = bodyweight (kg), C18:2 = the quantity of linoleic acid, C≥20 = the quantity of fatty acids with a chain length equal to or greater than 20 atoms of C, CEL = cellulose (kg/d), CF = crude fiber (kg/d), CP = crude protein (kg/d), DEI = digestible energy intake (MJ/d), DMI = dry matter intake (kg/d), DOMI = digestible organic matter intake (g/d), EDm = energy digestibility at maintenance (% of GE), EE = ether extract (fat, kg/d), GE = gross energy, GEI = gross energy intake (MJ/d), HC = hemicellulose (kg/d), MEI = metabolizable energy intake (MJ/d), MN = multiple of maintenance, NFE = nitrogen-free extract (kg/d), NSC = non-structural carbohydrate (kg/d), S_{ADFI} = silage ADF intake (kg/d), T_{ADFI} = total ADF intake (kg/d).

a. Intercept becomes 59 in Eq. 2 if the diet is based on corn silage.

b. Although Ellis et al. (2007) developed 14 equations for beef, 8 for dairy, and 10 for beef and dairy, the best-fitting equations are presented here.

Models for national inventories

The Intergovernmental Panel on Climate Change (IPCC), in its revised reference manual (IPCC 2006), outlined two methodologies—Tier 1 and Tier 2—to estimate CH_4 emissions from livestock enteric fermentation. A third methodology, Tier 3, is also recommended for countries with detailed information on animal and diet characteristics. The IPCC method is described in greater detail in the section on critical issues in accounting. The Food and Agriculture Organization of the United Nations (FAO) modified the IPCC Tier 2 methodology and developed a CH_4 emission module for its life-cycle assessment study of the dairy sector (FAO 2010). The equation developed was based on a digestibility-dependent Ym value (proportion of the animal's gross energy intake lost as CH_4) in which digestibility was inversely proportional to CH_4 loss per unit of energy intake (table 18 Eq. 24). In the United States, the Environmental Protection Agency refined the IPCC Tier 2 methodology by increasing the level of detail of livestock data and spatial and temporal characterization (U.S. EPA 2011). The agency estimated methane emissions with the Cattle Enteric Fermentation Model (CEFM), which calculates country-specific Ym values using model simulation and published data (Mangino, Peterson, and Jacobs 2003). The fundamental equations in CEFM are from the IPCC Good Practice Guidance Tier 2 approach (IPCC 2000). The main difference is that CEFM calculates cattle sub-populations on a monthly basis instead of an annual basis, resulting in increased levels of detail, such as definitions of livestock subcategories, livestock populations by subcategory, and feed intake estimates for the typical animal in each subcategory.

Bannink, van Schijdel, and Dijkstra (2011) developed a Tier 3 approach for the Netherlands that evaluates specific details of nutritional management because the underlying mechanisms of enteric fermentation are represented dynamically. The Tier 3 approach is based on a mechanistic model discussed in greater detail in the next section. The predictions of the Tier 3 approach more closely agreed with those of an older version of IPCC Tier 2 (IPCC 1997) than with the updated version (IPCC 2006). Uncertainty of predictions was approximately 15%, which is lower than the estimated 20% for the IPCC Tier 2 approach. The most likely sources of uncertainty were thought to be errors in estimating feed intake, the stoichiometry of volatile fatty acid production, and the acidity of rumen contents. Most developed countries are expected to move toward the Tier 3 approach using country-specific mechanistic models.

Other statistical (empirical) models

Several empirical equations to predict CH_4 production in cattle have been published since the 1930s and '40s (table 18). The equations in table 18 use a range of animal and dietary factors as covariates in prediction of CH_4 emissions. Some of the driving variables for prediction include dry matter intake (e.g., Eq. 1, 2, 10, 16, 22), live weight (Eq. 7 and 16), milk production (Eq. 7 and 9), proportion of forage (Eq. 12 and 21) and chemical composition of the diet (Eq. 5 and 15). Figure 1 summarizes the driving variables that influence enteric CH_4 emissions. More than 75% of the equations in table 18 use some measure of intake (e.g., dry matter, organic matter, digestible energy, gross energy) because 60%–80% of the variation in CH_4 prediction could be attributed to a measure of intake (Mills et al. 2003; Ellis et al. 2007). Whole-farm models (discussed below) and ration-formulation models typically rely on empirical models to predict enteric fermentation. For example, version 6.1 of the Cornell Net Carbohydrate and Protein System (CNCPS) uses the Mills et al. (2003) and Ellis et al. (2007) equations to predict enteric CH_4 emissions (Van Amburgh et al. 2010). However, independent data for evaluation of the models within the CNCPS framework are not available.

Mechanistic models

A mechanistic model analyzes the behavior of a system in terms of its components and their interactions (Thornley and France 2007). Dynamic, mechanistic models express the time variable explicitly. Several dynamic, mechanistic models that estimate CH_4 emissions have been developed (e.g., Baldwin 1995; Mills et al. 2001). These models require detailed dietary input and base CH_4 production estimates on the simulated hydrogen economy in the rumen. It is assumed that the hydrogen produced in the rumen from fermentation of soluble carbohydrate and protein to volatile fatty acids is used (1) to support rumen microbial growth, (2) for biohydrogenation of unsaturated fatty acids to saturated fatty acids, and (3) for production of glucogenic VFA (i.e., propionate and valerate). The remaining hydrogen is used solely and completely for methanogenesis in the reduction of CO_2 to CH_4 (Baldwin 1995; Mills et al. 2001; fig. 12). Two mechanistic models currently used for estimation of CH_4 emissions that were developed on the basis of Newton's first law of thermodynamics (MOLLY and COWPOLL) and a conceptual model based on the second law of thermodynamics are discussed below.

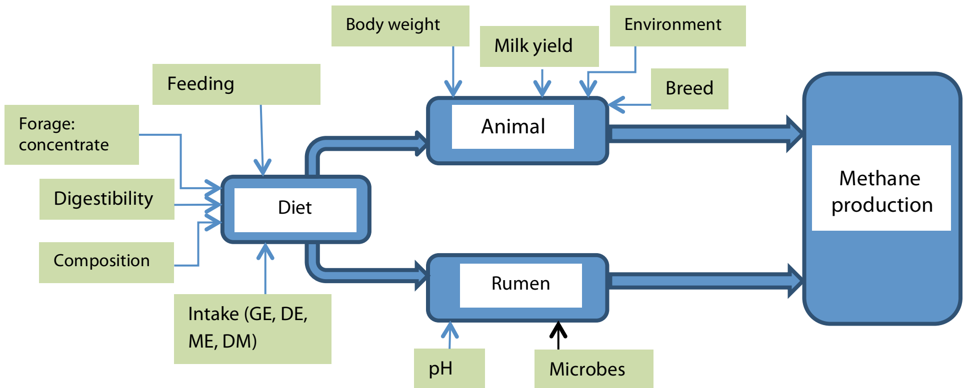

Figure 12. Factors affecting enteric methane emissions

MOLLY

A dynamic and mechanistic model developed at the University of California, Davis, MOLLY is based on dairy cows' rumen digestion and metabolism (Baldwin 1995). The model was constructed assuming continuous feeding, using Michaelis-Menten or mass action kinetics. The digestion element of the model is comprised of 15 state variables. The chemical composition of the diet is presented as starch, cellulose, hemicellulose, lignin, soluble carbohydrate, acetate, propionate, butyrate, crude protein (soluble and insoluble), non-protein nitrogen, urea, ash (soluble and insoluble), lipid, organic acid, lactate, pectin, and fat. After microbial attachment and substrate hydrolysis, the rumen model uses stoichiometric coefficients to convert starch, soluble carbohydrates, and amino acids into volatile fatty acids (VFA). The VFA stoichiometry is based on the equation developed by Murphy, Baldwin, and Koong (1982), which relates the amount of VFA produced to the type of substrate fermented in the rumen.

COWPOLL

The model developed by Dijkstra et al. (1992) served as a basis for development of the COWPOLL model. A dynamic and mechanistic model, COWPOLL is designed to simulate the digestion, absorption, and outflow of nutrients in the rumen. The model contains 17 state variables representing N, carbohydrate (fiber, starch, and sugar), lipid, and VFA pools. Chemical composition of the diet is presented as starch (soluble and insoluble), fiber (degradable and undegradable), crude protein (soluble and undegradable), water-soluble carbohydrate, ether extract, VFAs (acetate, propionate, butyrate and valerate), ammonia, ethanol, and lactate. Mills et al. (2001) added CH_4 production in the rumen and hindgut to the Dijkstra et al. (1992) rumen model. Kebreab et al. (2004) later integrated N transactions and, as such, developed an extended model. Given that VFA molar proportions are important determinants of CH_4 formation, COWPOLL uses a VFA stoichiometry, developed by Bannink et al. (2006), based on data collected from digestion trials with dairy cows. In addition to the stoichiometric differences described above, MOLLY and COWPOLL also differ in the number of microbial pools; MOLLY uses one microbial pool, whereas COWPOLL uses three pools (amylolytic, cellulolytic, and protozoa).

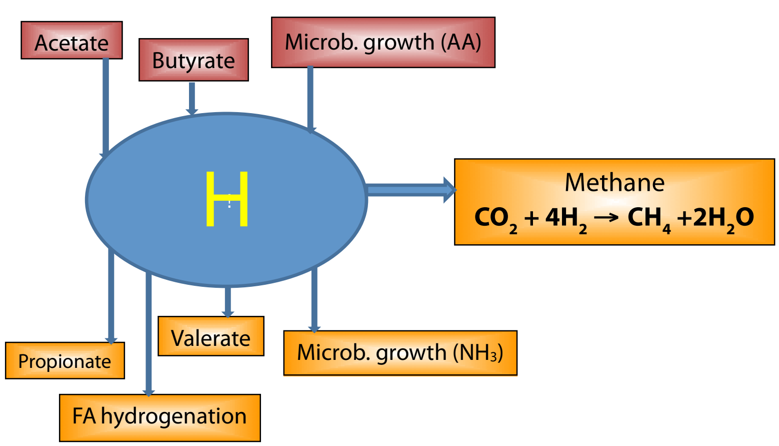

Figure 13. Schematic representation of the main driving variables for H_2 excess and methanogenesis in mechanistic models

Thermodynamic model

Kohn and Kim (2011) proposed an integrated thermodynamic and kinetic model to predict CH_4 emissions from VFA concentrations. The authors use organic matter digestion in the rumen to predict VFA production, absorption, and passage rates, which they in turn use to predict VFA concentrations. They use the second law of thermodynamics to explain why volatile fatty acid and CH_4, among other metabolites and gases, occur in the rumen and predict their quantities. The model is still at a conceptual stage and has not been tested with in vitro or in vivo data. Once fully developed and evaluated, the model might explain the mechanisms of some factors that change CH_4 emissions and might suggest ways to decrease enteric CH_4 emissions.

Whole-farm models

Many whole-farm models simulate the impact of various management or nutritional strategies on GHG emissions. Due to the relative simplicity of empirical prediction equations, a number of them are typically used in whole-farm models. The models' predictive power depends on the accuracy of these equations. Whole-farm models differ in their level of complexity and structure. Calculation software such as Cool Farm Tool[3] and Agricultural and Land Use National Greenhouse Gas Inventory[4] (ALU) are available online. These tools are mostly based on IPCC (2006) equations and simulate emissions from various types of animals, manure storage, and soil using a factorial approach, because nutrient flows from one phase of the farm (such as animal management) do not necessarily follow to the next phase. ALU has been developed with the help of U.S. EPA for inventory calculations in the United States. More complex models, such as Holos (developed by Agriculture and Agri-Food Canada) and the Integrated Farm Systems Model (IFSM, developed by USDA-ARS) simulate whole-farm emissions of GHG, including CH_4, and evaluate the overall impact of management strategies to reduce CH_4 emissions. The IFSM is a process-based (mechanistic) whole-farm simulation model that incorporates soil processes, crop growth, tillage, planting and harvest operations, feed storage, feeding, herd production, manure storage, and economics (Rotz et al. 2011). The IFSM uses the Mills et al. (2003) equation to predict CH_4 emissions from enteric fermentation. Independent evaluation of the enteric CH_4 emissions equation as used in IFSM is not yet available.

Ellis et al. (2010) evaluated empirical equations used to predict CH_4 emissions from dairy cows in eight whole-farm models. In general, predictions were poor; several models severely over-or under-predicted CH_4 production. These results suggest that the evaluated equations cannot fully describe the underlying causes of variation. Simple, more generalized equations performed worse than those representing some aspects of diet composition. Although some of the equations include basic aspects of the chemical composition of the diet and indirectly take into account the effect of feed intake level, they lacked other important descriptors that affect CH_4 production such as fat fraction (as reviewed by Beauchemin et al. 2009). Most of the equations also assume a constant CH_4 yield per unit of substrate (e.g., non-structural carbohydrate), but variation in intake level may affect their rumen fermentability and pH and, as a consequence, VFA profile and CH_4 release (Bannink et al. 2006). Although whole-farm models performed adequately with the inclusion of IPCC Tier 2 equation, the equation is based simply on gross energy intake and does not fully describe changes in composition of the diet (Ellis et al. 2010). For example, gross energy intake may increase due to greater feed intake, likely resulting in increased CH_4 emissions, but gross energy intake may also increase if the dietary fat content of the diet is increased, likely resulting in decreased CH_4 emissions. If a whole-farm model is attempting to describe the effect of a given mitigation strategy on CH_4 emissions, and subsequently the effects on other aspects of the farm nutrient balance, empirical equations may not be sufficiently accurate, because they are not adequately sensitive to change. The low prediction accuracy and poor prediction of variation in observed CH_4 production values may introduce substantial error into inventories of GHG emissions and lead to incorrect mitigation recommendations on a whole-farm level. Thus conservative estimates would need to be used. However, mechanistic models require more detailed input data that may not be readily available, hampering these models' inclusion in whole-farm models.

3. http://www.unilever.com/aboutus/supplier/sustainablesourcing/tools/.
4. http://www.nrel.colostate.edu/projects/ALUsoftware/index.html.

Comparison of models

Several studies have evaluated the prediction potential of empirical and mechanistic models for enteric CH_4 production from cattle using independent data sources (Benchaar et al. 1998; Kebreab et al. 2006a, 2008). Wilkerson, Casper, and Mertens (1995) reviewed several statistical models and recommended adoption of the Moe and Tyrrell (1979) equation for dairy cows. Mills et al. (2003) also compared various statistical models from the literature as well as models developed by the authors using extensive calorimetry data. Although the authors found that the Moe and Tyrrell (1979) model provided reasonable predictions, their own non-linear models not only improved predictions but also were less prone to misapplication. Linear models gave unrealistically high emission values as dry matter intake increased, whereas non-linear models yielded values approaching theoretical maximum emissions, a prediction that is biologically realistic. Benchaar et al. (1998) compared the predictive capacity of two mechanistic and two linear models with a database constructed from the literature. Predictions from linear equations were poor; the models explained 42% to 57% of the variation. The mechanistic models, on the other hand, explained more than 70% of the variation.

Kebreab et al. (2006a) chose six models, including two linear models (Moe and Tyrrell 1979 and Mills et al. 2003), a non-linear model (Mills et al. 2003), IPCC Tier 1 and 2 models (IPCC 1997), and a mechanistic model (COWPOLL) for predicting CH_4 production, and challenged them with North American data. The linear models were recommended for use where nutrient information is limited and within the range in which the models were developed. The non-linear model of Mills et al. (2003) could be used for extrapolation beyond the range of data used for developing the model, but the mechanistic model was recommended for assessment of mitigation options. The Tier 1 model was found to be adequate for general inventory of CH_4 emissions but inadequate for assessing the mitigation options, because it relies on default emissions factors. Although relatively less accurate in predicting the mean, the Tier 2 model is better suited when information on feed intake is available (Kebreab et al. 2006a).

Kebreab et al. (2008) evaluated two empirical models (IPCC 2006 and Moe and Tyrrell 1979) and two mechanistic models (COWPOLL and MOLLY) for their predictive capacity using individual cattle measurements of CH_4 emissions. In dairy cattle, COWPOLL yielded the lowest predictive error. However, in feedlot cattle, MOLLY yielded the lowest predictive error. The predictions of the IPCC model agreed in large measure with observed values. The average Ym values were 5.63% of gross energy (a range of 3.78% to 7.43%) in dairy cows and 3.88% (a range of 3.36% to 4.56%) in feedlot cattle. The authors reported that using IPCC values can result in overestimation of emissions by 12.5% for dairy cattle and underestimation by 9.8% for feedlot cattle. In addition to providing more accurate estimates of emissions based on diets, mechanistic models, unlike empirical models, can help researchers assess mitigation options such as changing carbohydrate sources and adding fat to reduce methane. Alemu, Ominski, and Kebreab (2011b) also compared empirical and mechanistic models to estimate and assess trends in enteric CH_4 emissions from western Canadian beef cattle. The authors concluded that a more robust approach may be to use mechanistic models to estimate regional Ym values, which are then used as input for IPCC models for inventory purposes. The CEFM model used by the U.S. EPA is an approximation of this approach but requires a mechanistic model to simulate Ym values in order to be adopted as a Tier 3 system similar to the Dutch methodology.

Critical Data Gaps Affecting Greenhouse Gas Emissions Quantification

To date, empirical models have been developed on the basis of relatively small amounts of data. The expense of measuring enteric fermentation makes collection of the vast amount of data necessary to establish relationships between animal and dietary inputs and CH_4 emissions challenging. Furthermore, the range of data collected is narrow, because most experiments are conducted to investigate treatment effects. Therefore, some of the data used to develop empirical models do not reflect animal and dietary attributes that may potentially be influential in predicting CH_4 emissions. For example, a number of equations in table 18 do not include ether extract (or fat), which is found to be influential in various studies. Ideally, empirical models should be developed from a database containing more than 1,000 records with detailed input parameters. Such a database would allow development of both general models, which use one or two covariates, and detailed models.

An important aspect of enteric CH_4 modeling is that the statistical method used in developing empirical models might not be appropriate. For example, most of the current models were developed using a frequentist statistical method. Using this method, a sequential application of simple significance tests can only be calculated. In addition, only nested models can be compared, and different models are selected if alternative procedures or starting covariates are included in the statistical procedures. To avoid methodological limitations, development of models in a Bayesian framework should be considered. One of the advantages of a Bayesian hierarchical methodology is that any combination of covariates is available, and each covariate is selected on the basis of the probability of that covariate being picked in a model during the analysis. In models developed in a Bayesian framework, animal and dietary variables can be made available for selection.

Fermentation in the rumen is a complex process involving microbial activities and degradable dietary components. Therefore, representation of this process using mechanistic models is also complex. In addition to degradation of dietary components and microbial growth, fermentation stoichiometry must be known to evaluate specific dietary components for the type of volatile fatty acid, H_2, and CH_4 produced during rumen fermentation. Alemu et al. (2011a) evaluated VFA stoichiometric models for their capacity to predict VFA molar proportion and CH_4 using independent data sources. They reported that variation among stoichiometric models in predicting VFA production had a major influence on the accuracy of estimated enteric CH_4 production. VFA stoichiometric models need to be improved to yield more accurate estimates of VFA production and absorption across the rumen wall.

Model Selection

Numerous models predict GHG emissions from livestock. Selection of the appropriate model depends on the user's objective (e.g., national inventory), data availability, and the relevance for various situations and management systems. Selection also depends on the limitations and uncertainty of models as well as the scale of operation, which affects the model's viability for accounting in GHG emissions reduction projects.

Development of models starts with an objective. Therefore, selection of a model should take into account the purpose for which the model was developed. Most empirical models were developed using limited data and may not be appropriate to extrapolate beyond the range of data from which they were developed. For example, a model developed on the basis of beef cattle data may not be appropriate for use in dairy cattle. However, empirical models may be preferable in some instances. Development of these models is often the first step in developing process-based models. Empirical models can be useful in conjunction with mechanistic models. One of the major constraints in using mechanistic models is data availability. Due to the complex chemical compositional requirements of livestock diets, it may be difficult to use mechanistic models for on-farm emissions quantification. Benchaar, Pomar, and Chiquette (2001) have demonstrated that mechanistic models are the superior, if not the only, option for comparing mitigating options using dietary manipulation.

5. CRITICAL ISSUES FOR GHG ACCOUNTING IN LIVESTOCK SYSTEMS

The Food and Agriculture Organization's report, Livestock's Long Shadow, raised concerns about the environmental impact of animal agriculture, particularly GHG emissions to the atmosphere (Steinfeld et. al 2006). Corporate social responsibility reporting has been enabled through guidelines set out by processes like the Global Reporting Initiative and Carbon Disclosure Project. The public and non-governmental sector as well as global food companies and their supply chains are developing tools to measure and manage GHG emissions and other aspects of on-farm sustainability. This effort builds on longer-term efforts to develop national inventories under the UNFCCC agreements using IPCC methodologies. The recent move toward project-level protocols for livestock GHG mitigation in Canada and the United States is an offshoot of the U.N. agreements as voluntary programs and regional regulatory programs for mitigation move forward.

Life-cycle assessment (LCA), typically based on ISO 14040/14044 standards, is one of the main accounting frameworks used to assess GHG and other metrics associated with the sustainability of supply chains for agricultural products (grower/producer – food manufacturing/processing sector – retail/restaurateur – consumer). LCA studies reveal that more than 70% of the GHG impact is attributable to the primary production and farm-level processing portion of the chain, shifting attention to measurement and management of on-farm sustainability (Hermansen and Kristensen 2011; Thoma et al. 2012; Thoma et al. 2011). Project-based accounting, based on the ISO 14064:2 or World Resources Institute GHG Protocol standards, is another carbon accounting framework used by GHG registries and programs for voluntary and compliance-based carbon offsets.

Various programs and registries apply these project-based accounting frameworks to develop program-level standards or guidance as well as carbon offset protocols for quantifying GHG reductions from management changes in the agricultural and other sectors. The accounting frameworks for GHGs under project-based quantification differ from LCA accounting in some major ways, a fact not sufficiently realized by GHG practitioners and accounting-framework end users.

As mentioned, several international standards guide product development under both frameworks (LCA accounting and GHG quantification protocols). However, the intravariability of products and results from the LCA accounting framework tends to be larger than that within project-based accounting frameworks, simply because the methodological choices are broader. Moreover, LCA accounting criteria are less clear; these criteria depend on objectives and methodological choices, whereas carbon offsets have a fairly consistent set of criteria

Differences in the measurement, monitoring, reporting, and verification systems to implement LCA accounting and project-based accounting frameworks are discussed below.

Quantification Frameworks

Many tools are used to quantify GHGs in agriculture (Driver, Haugen-Kozyra, and Janzen 2010), for different purposes (fig. 14). In an ideal world, the scientific basis or underlying quantification methodology for the four general classes of tools shown below would be the same. However, accounting and implementation frameworks, methodological choices, and application of the quantification methodology within each context differ significantly. The result is confusion about interpretation of results and skepticism on the part of the agricultural sector about the application and veracity of GHG values attributed to its operations.

Importance of IPCC guidance and approaches in livestock quantification

Increasingly, country-specific adaptation of IPCC guidance in national inventories is viewed as a valid scientific source of GHG quantification methodologies for use in LCA and project-based accounting frameworks. This acceptance is partially driven by the consistency principle in ISO and other process-based standard guidance.

The IPCC, in its revised reference manual (IPCC 2006), outlined two methodologies to estimate CH_4 emissions from livestock enteric fermentation, Tier 1 and Tier 2. A third methodology, Tier 3, is also recommended for countries with detailed information on animal and diet characteristics. Tier 1 is a simplified approach that assigns default CH_4 emissions for distinct animal categories. Therefore, only readily available animal population data are needed to estimate emissions. When more detailed livestock data are available, Tier 2 method estimates CH_4 emissions by using CH_4

emissions factors (Ym, table 18 Eq. 19). The Ym represents the proportion of the animal's gross energy intake lost as CH_4. Several countries and agencies have adapted the IPCC methods and customized Tier 2 approaches with country-appropriate Ym factors. Furthermore, several countries are beginning to apply Tier 3 methods through the use of mechanistic-based models to achieve a more dynamic approach to estimating enteric emissions. Most developed countries are expected to adopt the Tier 3 approach using country-specific mechanistic models. As shown below, both project-based and carbon footprint accounting frameworks are adopting IPCC-based approaches in the quantification estimates of farm-based emissions.

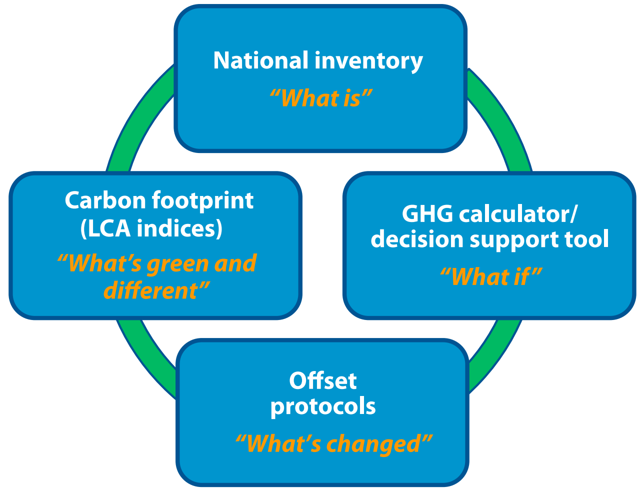

Figure 14. Quantification tools in agriculture

U.S. national GHG inventory

Quantifying methane from enteric fermentation and CH_4 and N_2O emissions from manure storage and handling is well characterized by IPCC best practice guidance (IPCC 2006). The science laid out in the IPCC guidance is applied in national emissions inventory reporting to the U.N. Framework Convention on Climate Change. The U.S. inventory applies IPCC Tier 2 methodology to estimate emissions from cattle (because CH_4 emissions from cattle are higher than from other types of livestock) and Tier 1 methodology for all other types of livestock (U.S. EPA 2011; USDA 2011a).[5]

Tier 1 IPCC methodology is based on default emission factors, but Tier 2 requires the user to calculate them using regionally specific information. These factors are estimated according to (1) animal category, which may include gender and life stage (e.g., nursing calves, heifer stockers, or feedlot steers in cattle operations; weaner pigs or feeder pigs in hog operations; lactating cows or replacement heifers in dairy operations), (2) each category's dry matter intake and feed ingredients (level of concentrates, total digestible nutrients, and crude protein); and (3) excretion of volatile solids and nitrogen in manure.

For cattle operations, eight major equations from IPCC (2006) are used to calculate emissions from the various relevant sources:

- CH_4 emissions from enteric fermentation (Eq. 10:21)
- concentration of nitrogen excreted for cattle (Eq. 10:22)
- daily volatile solid excreted for cattle (Eq. 10:24)
- CH_4 emissions from manure handling, storage, and land application (Eq. 10:23)
- direct N_2O emissions from manure (Eq. 10:25)
- direct N_2O emissions from manure storage (Eq. 10:25)
- indirect N_2O emissions from volatilization (Eq. 10:26)
- indirect N_2O emissions from leaching (Eq. 10:28)

Basic and applied studies directly investigating emissions from livestock operations in the United States and Canada are challenging these methods, particularly the equations' handling of the fat content and energy density of diets and certain default factors for GHG emissions from manure storage. These new studies suggest that Tier 1 methodology remains sufficient for emissions inventory but that Tier 2 methodology is more appropriate for use in GHG quantification in LCA or project-based accounting.

The IPCC guidance provides default assumptions that need to be tested against the literature review of CAST (2011b) and other emerging science. GHG emissions from land application of beef and dairy manure across the diversity of soils and cropping systems in the United States also requires additional research. In the U.S. inventory Tier 2 approach for cattle, the data needed to estimate the GHG emissions from the IPCC equations include

5. For a discussion of different IPCC tiers, refer to Olander et al. 2011.

- Beef and dairy cattle population variables for 10 subcategories:
  - USDA 2011b: cattle calf animal categories (age, gender) and by category each month (except bulls); cattle placed in feedlot by weight class; slaughter numbers and pregnancy/lactation data; mean weight gains/weight gains
  - USDA APHIS NAHMS 1997, 2008: calving percentages
- Beef and dairy diet variables (U.S. EPA 2010):
  - Regional diet characterization (State Livestock Specialists/APHIS NAHMS 2008): digestible energy (DE); energy requirements and fraction of energy converted to methane for each animal category and diet (Ym);
- Other data values (USDA 2010):
  - Extrapolated from above sources: body weight, net energy for activity, standard reference weight (dairy = 1,324 lbs; beef = 1,195 lbs); milk production; milk fat; pregnancy, DE and Ym.

The U.S. Tier 1 inventory approach for all other livestock uses population data from USDA NASS (1994) reports, updated with USDA Economics and Statistics System monthly, annual, and livestock population and production estimates. For horses, the FAOTSTAT database is used (FAO 2009). To estimate emissions, IPCC Tier 1 emission factors are applied per head of animal; no diet information or weights variables are required for these estimates (U.S. EPA 2010).

GHG calculators

Government departments, extension universities/colleges, and private sector companies have developed calculator-type tools for a variety of on-farm planning purposes (e.g., herbicide choice and application, fertilizer and nutrient application, and ration-balancing application).

In the GHG context, several model-based farm-scale calculators are available. These calculators include the COMET system, the DNDC calculator, Cool-Farm, and Holos (Olander et al. 2011; CAST 2011b). Typically, these calculators are based on IPCC emission factors (Tier 1 or 2) or on empirical and process-based models with user-friendly interfaces. Although the number of management practices and production scenarios tend to be limited and are based on the modeled parameters, these tools are increasingly being aligned with LCA and project-based accounting (e.g., by the National Pork Board and the Innovation Center for U.S. Dairy Sustainability initiatives). Other groups are incorporating financial data in the calculators to provide an economic analysis for adoption of GHG mitigation practices. This effort should increase the calculators' uptake by producers.[6]

Life-cycle analysis

LCA provides a snapshot in time of the environmental impact of a product or process, allowing that impact to be compared with the impact over time or of other products. Benchmarking over time using standardized LCA methodology can demonstrate progress in global environmental performance and can identify the specific processes generating strong environmental disturbances (hotspots).

Application of LCA to agricultural production systems and supply chains is relatively new. In the earlier 2000s, Walmart and several other companies successfully used LCA to design greener packaging and make shipped products more compact to reduce fuel use. However, agricultural production systems don't consume resources in a linear sense: they are interrelated cropping and animal systems and rely on renewable resources (e.g., nutrient recycling, seeds, cattle, and manure). These systems are based on biological systems with varying geographical conditions. Some simulation modeling is required to capture differences in these conditions.

Adapting LCA to agricultural production systems is not straightforward. In particular, livestock systems are influenced by abiotic factors such as agroclimatic drivers, inherent soil characteristics, and market drivers. Furthermore, agricultural production systems interact with many different sectors' materials and energy flows (fig. 15). In addition, cropping and animal production systems have multiple outputs or co-products that can have value or substitute as inputs in other production systems (e.g., meat, milk, hides, tallow, bone meal, manure, and other derivatives from dairy systems can be used in multiple production systems). Co-products require the LCA practitioner to apply allocation rules to partition the attribution of the environmental burden or benefit among the product system under study and the other production systems involved. Several methods have evolved to properly allocate environmental burden/benefit according to the co- or by-products, but this aspect of LCA application is certainly one of the more controversial and arbitrary in agriculture; harmonization of this

6. Pers comm., Belinda Morris, EDF CIG Grant on Demonstrating GHG Emissions from Rice Cultivation.

Figure 15. Materials and energy flow in agricultural production

aspect would increase certainty in LCA results. As the bio-economy and renewable energy sectors grow, capturing the true environmental impact of a production system through an LCA will become increasing challenging.

In a literature review, RIRDC (2009) found that most agricultural LCA studies used the ISO 14040:2006 and ISO 14044:2006 standards. These standards, like those used in project-based accounting, provide a consistent framework for conducting an LCA. However, that framework allows for flexibility in implementation of the accounting process, e.g., with respect to goal, system boundary, environmental impact categories to be included (e.g., air pollutants, GHGs, or water discharges or energy or land use), choice of functional units, allocation methods for co-products, primary and secondary data sources, and choice of LCA software and evaluation methods. Supply chain and product LCA standards have emerged to help facilitate comparison of agricultural LCA studies.

Project-based accounting

The ISO 14064:2 standard, or WRI GHG project-based accounting standard, allows for accounting of GHG emissions reductions from a baseline level after a GHG mitigation project has been implemented.[7] This approach is based on the systematic and comprehensive identification and analysis of baseline and post-project sources of GHG emissions, sinks, and reservoirs. The ISO 14064:2 standard provides a template and a process to (1) ensure that quantified emissions are based on a streamlined LCA for baseline and post-project conditions; (2) identify the relevant GHG emissions sources and sinks controlled by the project and sources and sinks that are upstream and downstream of the project as well as any leakage effects from the project activity (related and affected), and (3) identify GHG sources and sinks are material to or significantly affected by emissions reductions.

Project-based accounting standards are typically regime-neutral and can be adopted by companies, organizations, and governments to provide consistency and confidence in measurement, reporting, and verification (MRV) procedures (quantification protocols) in the context of a particular GHG framework. These standards cover project accounting details, including guidance on the gases to be counted and the methods for counting them, the different kinds of baseline scenarios and the procedures to assess the appropriate baseline, ways to address measurement uncertainties, data quality management, ways to set boundaries for measurement and monitoring, ways to address leakage of GHG impact outside those boundaries, and quantification of reversals. The standards also address requirements for monitoring, documentation, reporting, and verification of projects. Specific principles and requirements within a quantification protocol generated from these standards can be audited to ensure compliance. Only the GHGs sources and sinks that materially impact the quantification need to be quantified.

7. http://www.iso.org/iso/catalogue_detail?csnumber=38381 and the WRI GHG Protocol Standard for project accounting http://www.ghgprotocol.org/files/ghg_project_protocol.pdf.

Project-based accounting compares baseline emissions sources and sinks with post-project emissions sources and sinks, whereas LCAs strive to assess all emissions from sources and sinks of a product or process, either from cradle to grave or cradle to farm gate. The resulting GHG impacts or assessments from LCAs and project-based accounting are not readily comparable. Project-based methods only account for the relevant differences between baseline and post-project emissions changes, whereas LCA methods attempt to quantify the emissions profile of the product or process. Project-based accounting is used in carbon offset programs and registries.

Use of LCAs in Livestock Production Systems

Traditionally, LCA has been applied in the industrial sector on the basis of the material and energy flows of engineered systems and has been aimed at assessing the environmental impacts of relevant input and outputs of a manufacturing product or process. Classical attributional LCAs (ALCAs) relate these impacts to a functional unit, or the main function of a production system in quantitative terms. Adapting LCA to agricultural production systems at the primary level has proved to be a challenge.

According to the ISO LCA standards mentioned above, the four phases of an LCA study, with various methodological choices embedded in each, are (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation.

Phase 1: Goal and scope definition

The scope of an LCA, including the system boundary and level of detail, and to some degree the functional unit, depends on the study's subject and intended use. The depth and breadth of LCAs can vary considerably, depending on a study's goal and scale. In phase 1, life-cycle stages and environmental impact categories are identified. The practitioner must determine the type of allocation method to be used and decide whether to apply a consequential LCA or an attributional LCA. Attributional LCAs quantify the environmental impact of a certain amount of the product (i.e., functional unit in a status quo situation), whereas consequential LCAs quantify the environmental consequences of a change in the demand for the product (Thomassen et al. 2008). In reviewing 16 European livestock product LCA studies, de Vries and de Boer (2010) found that the common practice was to apply the attributional LCA.

Figure 15 indicates some primary (Scope 1) and secondary (Scope 2) emissions sources in an LCA, but a cradle-to-grave assessment requires consideration of tertiary (Scope 3) sources (Lal 2004). Tertiary sources include emissions from acquisition of raw materials, fabrication of tractors, operation of equipment and buildings, manufacture of packaging, product/commodity transport to markets, and indirect land use changes (changes in land use elsewhere attributed to the production system). Hermansen and Kristensen (2011) identified the various life-cycle stages of livestock products in the supply chain (fig. 16). Varying study scopes and system boundaries make direct comparisons among studies problematic. In practice, the majority of agricultural LCA studies designate the system boundary as cradle to farm gate (RIRDC 2009; de Vries and de Boer 2010) rather than cradle to grave, which would include transport and processing, packaging, and retailing downstream from the farm.

Regarding allocation of co-products, three main allocation methods can be applied: economic (by value of the co-product), physical (by mass of the co-product), and system expansion (by expanding the production system to include alternative production pathways using co-products) (ISO 2006). In a review of a wide variety of agricultural LCA studies, RIRDC (2009) found that the order of preference for allocation was system expansion, followed by physical, followed by economic value. However, in a review of 16 European livestock LCAs, de Vries and de Boer (2010) found that allocation by economic value was much more common. Several authors acknowledge that for livestock systems, economic value allocation can lead to greater uncertainty, particularly with beef and dairy products, because of their integrated co-product systems (i.e., allocation of meat and milk from the dairy sector; RIRDC 2009; Vergé et al. 2011).

Phase 2: Inventory analysis

The life-cycle inventory analysis (LCI) phase is an inventory of input and output data with regard to the system of interest. It involves collection of the data necessary to meet the goals of the defined study. In traditional industrial applications, life-cycle inventory databases for these sectors are relatively well established, bringing a level of harmonization in approach. However, the state of LCA practice in agriculture is inconsistent and non-standardized; some of the underlying LCI databases that support the analysis have international default factors, while others have national default factors or simply have data gaps when applied to primary agricultural production. Most practitioners and policy

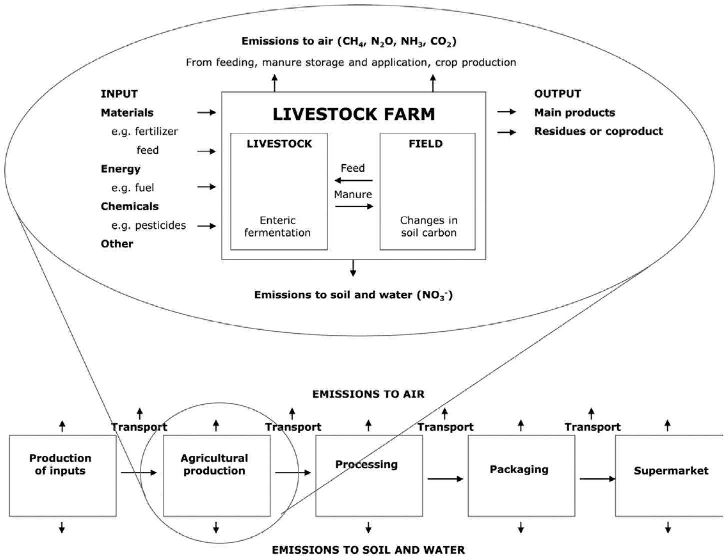

Figure 16. LCA life-cycle stages for livestock products in the supply chain
Source: Hermansen and Kristensen 2011

makers recognize that LCI inventory datasets should reflect local, primary data sources from the sector or commodity being modeled to improve LCA analysis and modeling. Simulation modeling for some of the more biologically based emission sources may be needed. Secondary sources of emission factors—such as fossil fuel extraction and processing, electricity production, and the production of fertilizer, medicines, and pesticides—would need to be derived from existing LCI databases, national- or state-level inventories, or industry input.

Phase 3: Impact assessment

The life-cycle impact assessment (LCIA) phase of the LCA provides additional information to help researchers assess a product system's LCI results to better understand their environmental significance.

Phase 4: Interpretation

The interpretation phase is related to the goals of the LCA. In reviews to date, the majority of the agricultural LCA studies were contributional and hotspot analysis, with a focus on changing feed choices or farming practice, rather than contribution and comparative analysis across different production systems (RIRDC 2009).

In summary, the main challenges in applying LCA accounting to GHG emissions from livestock production systems are differences in

- Goals and scales of LCA analyses
- Scope and boundary of LCA applications
- Functional units (e.g., kg CO_2e can be reported on kg live weight leaving the farm gate; kg of carcass weight; kg of retail cuts; gallon of milk, gallon or kg of fat-corrected milk, gallon or kg of fat- and protein-corrected milk; kg of protein; kg of energy; and kg of human non-edible feed, among others)
- Allocation methods associated with co-products of livestock or crop and feedstuff production
- Integration of livestock production with the surrounding environment and landscape (e.g., nutrients arising from livestock production can move beyond the farm boundary either naturally or by man-made means)
- Configurations of feeding and production systems (e.g., ruminants versus monogastrics as well as barn/feedyard types; choice of feedstuffs)
- Input-to-output ratios of agricultural production (e.g., yields can vary from year to year depending on rainfall or extreme weather events)
- Nature of GHG emissions from livestock production systems, precluding direct measurement and necessitating modeling and other prediction techniques
- Regional and geographical influences on feed production and manure emissions
- Primary and secondary data sources
- LCI databases for emissions source factors in agriculture

For these and other reasons, the carbon footprint of livestock products can vary significantly. For example, when comparing beef LCA studies, the range of GHG emissions can be as low as 10.4 kg CO_2e of beef produced to more than 80 kg CO_2e per kg of beef produced (Vergé et al. 2008; Leip et al. 2010). Table 19 presents the carbon footprint found in North American beef, milk, and pork studies.

Table 19. LCAs of North American pork, milk, and beef production systems

Study	LCA goal	LCA purpose	Scope/system boundary	Functional unit/kg CO_2e/ Intensity metric	Allocation method	Data quality/ variability	ISO 14040/ 14044 compliance	LCA evaluation
Pork								
Vergé et al. 2008	Understand C intensity of the Canadian pork sector (CLCA)	Compare C intensity change from 1981 to 2001	Cradle to farm gate (cradle meaning cropping and energy for feed production, upstream fertilizer/pesticide production)	2.99 kg CO_2e/kg live wt in 1981 2.31 kg CO_2e/kg live wt in 2001 (3.1 kg CO_2e/kg carcass wt in 2001)[a]	Not done – ended at farm gate (manure could have been allocated)	No assessment of uncertainty; data mostly secondary; some farm data or primary data gathered	Unstated; IPCC Tier 2-based models for CH_4 and N_2O estimates; cropping complex defined by region	Contribution and comparative analysis over time; data gaps for on-farm energy use recognized
Thoma et al. 2011 U.S. pork industry C footprint as part of Sustainability initiative	Scan Level (SLLCA) – understand supply chain C intensity Detailed LCA (DLCA) – focus on farm-level C intensity	SLLCA – understand pork production contribution to supply chain GHGs DLCA – understand baseline level of GHGs to support voluntary C markets	Cradle to grave; upstream crop production inputs as above; on-farm pork production; transport; processing; packaging; distribution, retail, and solid and liquid waste (even post-consumer)	4 oz (uncooked) serving of boneless pork for analysis Reported: 2.8 kg CO_2e/kg live wt 3.8 kg CO_2e/kg carcass wt	Economic value as base case (given a scan level) and mass allocation for feed byproducts	Monte Carlo assessment for knowledge and variability uncertainty; data mostly secondary from well-established sources; industry input for post farm-gate sources	Yes; Computational farm model based on productive life of one sow; utilized IPCC Tier 2 methodology for CH_4 and N_2O emissions	Contribution and hotspot, and comparative as some comparisons of manure mgmt. were done; two critical reviews conducted
Beef								
Capper 2011	Compare efficiency gains of modern beef production with 1977 levels	Highlight improvements in efficiency; identify opportunities for mitigation in future years	Cradle to farm gate, including cow-calf, stocker, feedlot and dairy (culled cows); primary and secondary inputs[b]; GHGs, water, nutrients, land	21.44 kg CO_2e/ kg carcass wt/yr 1977) 17.94 kg CO_2e/ kg carcass wt/yr (2007)[c]	Not stated, but appears to be mass or biological, based on input/outputs for culled dairy cows; ends at farm gate	No assessment of uncertainty; mostly secondary data from well-established national/state/ academic sources; some industry input gathered	Unstated; applied deterministic model (Capper 2009) applying resource inputs/ waste outputs	Contribution and comparative analysis
Beauchemin et al. 2010	To estimate whole-farm GHG emissions in western Canada	Determine relative contributions of cow-calf and feedlot and emissions attributable to CH_4	Cradle to farm gate; feed production; manufacture of fertilizers/ pesticides; on-farm fuel use; CH_4, N_2O, and fossil CO_2 over 8-yr period	22 kg CO_2e/kg carcass wt cow-calf – 80% of emissions feedlot – 20%	Not done; based on 120-head herd; representative cropping; beef production and native prairie grazing farm; ends at farm gate; no culled dairy calves	No stated assessment of uncertainty, however, Holos does calculate it; mostly secondary data from well-established national/prov'l/ academic sources	Unstated; used Holos – empirical model based on IPCC Tier 2 methodologies customized to Canada for all GHG emissions	Contribution analysis; hotspot identification – suggested areas for more research on mitigation
Beauchemin et al. 2011	Compare potential GHG mitigation strategies for beef production in western Canada	Using methodology in above study, identify GHG mitigation strategies from base case	See above; assessed feeding strategies (altered forage; added fats, corn DDGS, and forage quality); improved husbandry of breeding stock; reproductive performance	Base case – 22 kg CO_2e/kg carcass wt cow-calf herd – up to 8% reduction per strategy; up to 17% reduction together feedlot – up to 4% if applied together – overall up to 20%	See above; manure could have been allocated; other system's byproducts used as inputs such as fats and corn DDGS could have been allocated	No stated assessment of uncertainty; noted that soil C sequestration in well-managed pastures can tip balance from a net source to a net sink – but highly variable	See above	Comparative analysis; mitigation strategies compared
Vergé et al. 2008	Understand GHG emissions from beef production in western Canada	Compare C intensity change from 1981 to 2001	Cradle to farm gate (cropping and energy for feed production, pasture, upstream fertilizer/pesticide production, on-farm energy use, meat production)	16.4 kg CO_2e/kg live wt in 1981 10.4 kg CO_2e/kg live wt in 2001 (16.0 kg CO_2e/kg carcass wt in 2001)[d]	Not done – ended at farm gate (manure could have been allocated; no culled dairy animals considered)	No assessment of uncertainty; data mostly secondary; some farm data or primary data gathered	Unstated; IPCC Tier 2-based models for CH_4 and N_2O estimates; cropping complex defined by region	Contribution and comparative analysis over time; data gaps for on-farm energy use recognized

Study	LCA goal	LCA purpose	Scope/system boundary	Functional unit/kg CO_2e/ Intensity metric	Allocation method	Data quality/ variability	ISO 14040/ 14044 compliance	LCA evaluation
Pelletier et al. 2010	Compare beef production systems in the upper Midwest United States	Representative 100 cow-calf herd; GHGs, energy; environmental footprint and nutrient emissions for feedlot, feedlot/ back-grounding, and pasture finishing	Cradle to farm gate; cropping and energy for feed production, pasture; on-farm energy use, meat production (uncertain if secondary inputs included)	14.2 kg CO_2e/kg live wt (feedlot) or 21.8 carcass 16.2 kg CO_2e/kg live wt (feedlot/ back-grounding) or 24.9 carcass 19.2 kg CO_2e/kg live wt (pasture) or 29.5 carcass forage utilization rate (CO_2e/kg live wt): 30% – 21.5 60% – 19.2 90% – 18.4[e]	Gross chemical energy content of co-product streams (for feed energy)	No assessment of uncertainty Data mostly secondary; however, efforts were made to gather as much primary data from beef operations as possible	Yes; GHG emissions based on IPCC Tier 1 for most sources; Tier 2 for enteric methane; Ym of 5.5% used rather than 3% for concentrate diets (sensitivity analysis performed on both)	Contribution and comparative analysis; recognized that pasture C sequestration may have altered the results
Basarab et al. 2012	Understand GHG emissions from 4 beef production systems in western Canada	Compare the C footprint of calf-fed no implants (CFNI) with implants (CFI) yearling-fed no implants (YFNI), and with implants (YIF) in 4 actual herds w/associated crop and pasture production	Cradle to farm gate; cropping and energy for feed production, pasture, upstream fertilizer/pesticide manufacture, seed processing, and transport; on-farm energy use, meat production; also adjusted for time for efficiency comparisons	21.09 kg CO_2e/kg carcass wt CFNI 22.60 kg CO_2e/kg carcass wt/y CFNI 19.87 kg CO_2e/kg carcass wt CFI 21.28 kg CO_2e/kg carcass wt/y CFI 22.52 kg CO_2e/kg carcass wt YFNI 39.31 kg CO_2e/kg carcass wt/y YFNI 21.21 kg CO_2e/kg carcass wt YFI 37.02 kg CO_2e/kg carcass wt/y YFI	Economic allocation for straw; no other byproduct allocation	No assessment of uncertainty; almost all data from primary on-farm sources except for some secondary upstream inputs	Yes; CH_4 and N_2O emissions based on IPCC 2006 Tier 2 methods; 4 actual beef production farms used in calculations.	Contribution and comparative analysis; recognized that pasture C sequestration may have altered the results
Dairy								
Vergé et al. 2007	Understand GHG emissions from milk production in Canada in 2001	Quantify carbon footprint in 2001 per animal and as a function of milk production	Cradle to farm gate (cropping and energy for feed production, pasture, upstream fertilizer/pesticide production; on-farm energy use, milk production for 5 regions)	4.55 t CO_2e/live wt 1.0 kg CO_2e /kg milk	Not done – ended at farm gate (but acknowledged dietary fat additions, imported manure, and culled cows could be allocated	No assessment of uncertainty; data mostly secondary; some farm data or primary data gathered	Unstated; IPCC Tier 2-based models for CH_4 and N_2O estimates; cropping complex defined by region	Contribution and comparative analysis
Dyer et al. 2008	Understand trends in GHG emissions from milk production in Canada	Compare carbon intensity change from 1981 to 2001	Based on above scope and study method, but with an indexing method to the original IPCC calculations	1.22 kg CO_2e /kg milk (1981) 0.91 kg CO_2e /kg milk 2001)	Same as above	Same as above	Same as above; index method	Contribution and comparative analysis over time
Capper, Cady, and Bauman 2009	Understand trends in environmental impact of milk production in the United States	Compare environmental efficiency gains in milk production from 1944 to 2007	Cradle to farm gate; cropping and energy for feed production, pasture, upstream fertilizer/pesticide production; on-farm energy use, milk production	3.66 kg CO_2e /kg milk (1944) 1.35 kg CO_2e /kg milk (2007)	None; although acknowledged use of byproducts for feed/fiber would reduced the footprint	No assessment of uncertainty; mostly secondary data from well-established national/state/ academic sources; some industry input gathered	Unstated; applied deterministic model (Capper 2009) applying resource inputs/ waste outputs; IPCC based for CH_4 and N_2O emissions	Contribution and comparative analysis over time
Thoma 2012	Understand the carbon intensity of the fluid milk supply chain for sustainability planning	Apply a scan-level analysis to identify where emissions occur along the chain						

Study	LCA goal	LCA purpose	Scope/system boundary	Functional unit/kg CO_2e/ Intensity metric	Allocation method	Data quality/ variability	ISO 14040/ 14044 compliance	LCA evaluation
Rotz et al. 2010	Develop a dairy model for estimating C footprints for U.S. dairy production systems	Apply the model to a variety of production systems; sensitivity analysis	Cradle to farm gate (cropping and energy for feed production, pasture, upstream fertilizer/ pesticide, fuel, machinery, and plastic production; on-farm energy use, emissions from off-farm replacement animals; milk production	PA farms: 0.53 kg CO_2e/kg ECMf milk (500 cow confined) 0.46 kg CO_2e/kg ECM milk (2000 cow drylot) CA farms: 0.57 kg CO_2e/kg ECM milk (500 cow confined) 0.47 kg CO_2e/kg ECM milk (2000 cow drylot)	Economic allocation between milk and meat in dairy production	Sensitivity analysis revealed enteric methane most sensitive, but a 10% error still caused only a 6% variance in C footprint; appears data sources are primary for on-farm production	Yes; DairyGHG model derived from Integrated Farm System Model (IPCC-based); included C sequestration using COMET-VR	Contribution and comparative analysis; sensitivity analysis; verification of model against published studies

a. Based on using a conversion of 0.75 lb pork carcass per lb of live weight hog.
b. Primary: feed production, water, energy use, transport emissions; secondary: fertilizer/pesticide manufacturing, fuel for cropping practices, irrigation water.
c. More interesting is the reported "reduced maintenance effect over time" from increased growth rates in steers and shortened days to harvest in the 2007 beef production systems. The reduction in total feed energy for maintenance per 109 kg of carcass wt of beef went from 251,090 x 106 MJ in 1977 to 230,898 x 106 MJ in 2007, reflecting a greater proportion of calf-fed beef and dairy steers entering the feedlot and fewer animals on pasture and in stocking phases.
d. Based on a conversion of 0.65 kg of beef carcass per kg of live weight animal.
e. The authors note that if C sequestration in pastures were accounted for (based on published values), the difference between well-managed pasture finished beef and feedlot finished beef would decrease by 6%.
f. Energy-corrected milk (ECM) is based on 3.5% fat and 3.1% protein concentrations.

Comparisons of carbon footprints for different livestock types can be misleading, but a number of researchers have provided some general observations (de Vries and de Boer 2010; RIRDC 2009; Vergé et al. 2011). In North America, comparative evaluations of LCA tend to focus on either reduced environmental impact over time, using the same LCA methodological approach (Capper, Cady, and Bauman 2009; Capper 2011; Vergé et al. 2007 and 2008; Dyer et al. 2008), or comparing production systems within a livestock type using the same methodological approach (Pelletier et al. 2010; Beauchemin et al. 2011; Basarab et al. 2012; Rotz et al. 2010).

In a review of LCA studies in livestock production systems (Hermansen and Kristensen 2011; RIRDC 2009), the following observations were made:

- In an average OECD diet (de Vries and de Boer 2010), differences among the environmental impacts of a kg of pork, chicken, and beef are explained by differences in (1) feed efficiency and feed production types, (2) enteric CH_4 emissions of ruminants and those of monogastrics, and (3) reproductive rates that impact productivity and production period (e.g., in pork and chicken production, the level of GHGs from breeding stock is relatively low due to a relatively large number of progeny per mother animal annually)
- Further, Hermansen and Kristensen (2011) point out that differences in LCAs can arise due to (1) the way in which or degree to which livestock production systems are integrated into a region's land use and the choice of methods to account for indirect land use change and (2) how manure substitution for synthetic fertilizers is accounted for in the LCA methodology.

These authors acknowledge that three factors are important in reducing the GHG footprint of livestock products: mitigation strategies that reduce emissions from (1) feed use, (2) manure handling, and (3) feed production (i.e., reduce the carbon footprint of the feed produced).

Mitigation strategies for feed production are covered in significant detail in companion reports (Olander et al. 2011; Eagle et al. 2012).

Use of Project-Based Accounting in Livestock Production Systems

Most—if not all—formalized carbon markets, whether voluntary or regulatory, use approved GHG quantification protocols to streamline project development risk and costs, facilitate third-party verification, and provide certainty to the marketplace. These standardized protocols are largely based on the principles and approach laid out in the ISO 14064:2 standard. They are typically developed with a view to balancing specificity for comprehensive GHG accounting for a project type and flexibility to apply to various configurations for the project type identified by the protocol. In addition to the ISO 14064:2 process-based standard, the overall policy and market frameworks of the programs or registries (as laid out in program guidance documents or program standards) guide the application of these protocols to ensure that the offsets meet the requirements of the relevant market.

A quantification protocol allows project developers to save costs and reduce risk by ensuring that GHG reductions or removals quantified from a particular project type meet the eligibility rules and policy criteria of the given program or registry. A protocol provides clarity, consistency, and confidence in measuring (or estimating), monitoring, reporting, and verifying emission reductions. The result is enhanced transparency, integrity, and credibility of the carbon market and resulting carbon offsets for sale in the marketplace.

Protocols can be developed in a top-down process by the regulator or program manager (e.g., Climate Action Reserve, Alberta Offset System) or a bottom-up process in which prospective project developers must first get approval for a draft protocol (e.g., Verified Carbon Standard, VCS, or Clean Development Mechanism, CDM). Some programs allow both top-down and bottom-up protocol development and approval processes (e.g., American Carbon Registry). In each case, development of these protocols is commonly associated with significant human and financial investments in scientific and best-practice knowledge, industry knowledge, facilitation, review, public scrutiny, approval, and assurance of applicability to the policy criteria of the given program or registry.

Protocol development has evolved to be more in line with the ISO 14064:2 standard. The ISO approach is based on the systematic and comprehensive identification and analysis of the sources of GHG emissions, carbon sinks, and carbon reservoirs. The ISO 14064:2 standard lays out general principles of relevance, completeness, consistency, accuracy, transparency, and completeness that form the basis for identification of these sources, sinks, and reservoirs (SSRs) and accounting choices in project-based protocols. ISO 14064:2 also provides a template and a process to ensure that quantification protocols (1) are based on a streamlined LCA for project and baseline conditions and identify all inputs and outputs analogous to LCA primary-, secondary-, and tertiary-scope analysis (i.e., controlled, related, and affected SSRs); (2) evaluate all potential baseline scenarios; (3) identify the relevant GHG emissions controlled by the project as well as impacts upstream and downstream of the project; and (4) decide which GHG sources and sinks are material to or significantly affect quantification methodologies.

The scope of development and review for protocols and project and baseline conditions can vary among systems. The required streamlined life-cycle assessment for GHGs in the ISO 14064:2 standard process for controlled, related, and affected SSRs has its advantages. One is that this systematic approach reduces the chance that any major positive or negative effects (or leakage) are overlooked. Nonetheless, a challenge for protocol developers is deciding the extent to which the downstream and upstream assessment should occur. That call is made on the basis of GHG effects that are material and can be expected to influence conclusions about net emission reductions in a non-conservative direction. This scope of analysis for protocols can vary greatly among systems, affecting other criteria such as leakage and uncertainty. Generally speaking, the grander the scope, the greater the difficulty in identifying and quantifying effects that are considered far upstream and downstream and thus under no reasonable control by the project. However, the key difference between the ISO 14040/14044 and ISO 14064:2 standard approaches is the relative comparison of baseline with post-project GHG sources, sinks, and reservoirs in the ISO 14064:2 standard approach. This comparison allows for quantification of only relevant GHG sources in lieu of mandatory quantification of all life-cycle impact activities.

In regulated markets, the regulator typically provides guidance on the scope of the review. For example, the regulator may determine that quantifying emissions from off-shore steel fabrication is unnecessary. This finding should lead to clear and achievable quantification procedures, in addition to consistency among all protocols and, subsequently, offset quantification. But, in voluntary programs, particularly those with an international scope, guidance, particularly in CDM and VCS systems, is more open.

The CDM may be the only notable exception to the move toward ISO 14064-2 compliance, but it was the pioneer in carbon offset markets and was implemented before the more common GHG quantification process standards were developed. Following a standard framework like ISO provides transparency to protocol development and helps to ensure consistency in GHG measuring, monitoring, reporting, and verification for projects (fig. 17); however, ISO provides only general guidance and leaves to protocol developers most decisions about how to translate the general ISO principles into practice for the particular type of GHG reduction project under analysis. As per figure 17, the quantification protocols incorporate a wide range of data, methodology, and guidance sources.

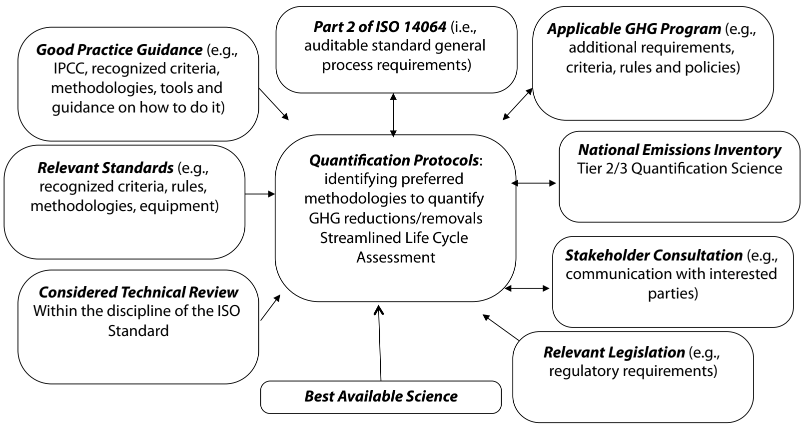

Figure 17. Standard framework for protocol development offered by the ISO 14064:2 process (ISO 14064:2 2006)

The protocols and their embedded quantification methods are increasingly moving toward an in-depth grounding in the relevant peer-reviewed scientific literature as well as national and international best practice guidance (most notably, the IPCC methodologies). This trend puts increasing responsibility on protocol developers to demonstrate a thorough scientific analysis of the net impact of all GHGs on site as well as upstream and downstream from the site that may be affected by the project, ensuring that the quantification methodology conforms to varied and evolving sources of information. This task can be challenging for the livestock production sector and other areas in which the science is not yet at a synthesis stage or quantification approaches vary. Consequently, broad consultation with science, technology, and GHG quantification experts is increasingly valued and needed to develop consensus on protocol quantification approaches, a key tenet of the ISO 14064:2 process.

The level of activity in livestock GHG mitigation protocols is far less than that of other agricultural activities for which protocols have been developed, with the exception of CH_4 capture and destruction from manure anaerobic digesters.[8] Feeding management strategies to reduce enteric CH_4 emissions from ruminants, feed additives (e.g., ionophores, lipids, beta-agonists) or animal husbandry techniques and other types of livestock management activities are less common (table 20). The Alberta Offset System has developed and approved several relevant livestock quantification protocols. Other livestock GHG mitigation protocols are in development, though not yet approved by programs/registries; some of these are included in table 19.

Table 20. Livestock protocols developed by program and registries

Protocol/initiative	Emissions scope	Status
Alberta Offset System* Dairy Cattle Emissions Reduction	Enteric CH_4, manure CH_4, N_2O	Approved
Innovative Feeding of Swine and Storing and Spreading of Swine Manure	Manure CH_4 and soils N_2O	Approved
Beef Feeding – Edible Oils	Enteric CH_4	Approved
Beef Reduced Days on Feed Beef Reducing Age to Harvest Selecting for Low Residual Feed Intake in Beef	All 3 protocols: enteric CH_4, manure CH_4, N_2O	Approved Approved Pending
Modification of Alberta Offset System Livestock GHG protocols for U.S.	Enteric CH_4, manure CH_4, N_2O	In development (ACR)
Development of a Modular Methodology for Quantification and Measurement of GHG Emission Reductions from Rumin ant Livestock Production	Enteric CH_4, manure CH_4, N_2O	In development (ACR)
Soil Carbon Sequestration through Rangeland Management	CO_2, CH_4, N_2O	In development by Environmental Defense Fund, UC Berkeley, and other organizations

* See www.carbonoffsetsolutions.ca

8. Anaerobic digesters as a mitigation activity are outside the scope of this report. The majority of programs and registries, including Climate Action Reserve, Air Resources Board, American Carbon Registry, Regional GHG Initiative, Alberta Offset System, and Verified Carbon Standard, have some form of CH_4 capture and destruction anaerobic digester protocol for manure and co-digestion products. Anaerobic digesters have been identified as an eligible project type under California's AB32 Cap-and-Trade Regulation and under the Western Climate Initiative.

Developing agricultural protocols under the Alberta development process requires significant coordination of relevant scientific research and technical data related to the GHG reduction or removal activity and baseline approaches. This coordination takes the form of technical seed documents, or TSDs.[9] The TSDs are working documents that identify the potential practices and technologies that will lead to the emission reductions and removals. They draw on best practice guidance to identify relevant activity data, emissions factors, and formulae to arrive at quantification approaches. TSDs should contain the most recent and relevant science from well-established sources and should ratify the link between practice change or new technologies and quantified GHG reductions. Typically, experts decide collectively on the synthesis science and technical issues under the discipline of the ISO 14064:2 process. As a result, the GHG quantification methodologies embedded in the Alberta series of livestock quantification protocols apply Tier 2 IPCC quantification approaches for estimating enteric methane and manure-based CH_4 and N_2O emissions.

Agricultural protocols typically develop through phases of collective decision making about and peer review of the certainty of the science at hand. The more complex the protocol, the longer it takes and agricultural protocols can take the longest depending on the availability and robustness of consensus and synthesis science (average length until approval is over 24 months). Table 21 shows the kinds of emission reduction activities that were finally accepted as providing real, measurable and verifiable reductions (see discussion in Validation/Verification section below) and their potential reductions in the Alberta Protocols.

Table 21. Reduction strategies/potential in livestock reduction protocols

Strategy/protocol	Enteric methane potential	Manure potential	Additional considerations
Adding edible oils 4% to 6% of dry matter in the diet	Suppresses methanogenesis – up to 0.29 tonnes CO_2e/hd/yr	Corn DDGS as a fat source must be weighed against increased N excretion and N_2O emissions	High cost of oils/lipids at this time is a barrier Beef tallow likely business as usual
Reducing age to harvest – less time spent on forage maintenance diets (i.e., stocker phases)	Reducing age to harvest by 3 months – up to 0.75 tonnes CO_2e/head/yr	Less manure excretion – up to 0.25 tonnes CO_2e /head;	Savings in production costs
Reducing days on feed in the feedlot ↑ Production efficiency (feed management)	Reducing days on feed by 7 days – up to 0.02 tonnes CO_2e /head/yr	Less manure excreted – up to 0.02 tonnes CO_2e /head	Feed costs reduced; less manure handling; individual animal performance; use of additives (e.g., beta-agonists)
Swine management ↑ Production efficiency (including feed strategies to reduce N and VS* excretion) Emptying basins to avoid CH_4 emissions Change timing/method of manure application	N/A	Empty manure storage in spring and in fall – up to 0.036 tonnes CO_2e /head/yr Manipulate feed rations – 0.025 tonnes CO_2e /head/yr	Feed costs reduced Changes in timing of manure application included in manure storage emptying
Dairy Management ↑ Milk productivity/feed management Emptying basins to avoid CH_4 emissions	Up to 0.5 tonne/head CO_2e/ head/yr)	Included in potential reduction to left	Use of additives like ionophores, lipids, forage quality, Prosilec Decrease use of replacement heifers, increased lactation cycles

* Volatile solids

For each of the emissions reduction mechanisms identified in the protocols, a credible baseline condition must be established. The baseline condition is assessed on the basis of the governing principles of the standard employed (i.e., ISO 14064:2). The baseline condition represents the most probable condition in the absence of the project. The applicable baseline condition can be assessed on a project-specific basis or as part of a broader approach, commonly termed performance standards.

Under a project-specific approach, each project must assess and substantiate an appropriate baseline, with guidance provided in the applicable protocol. Under the performance standard approach, a baseline that is broadly applicable across a given geography or within some other boundary is used for all applicable projects. The two types of baselines have advantages and disadvantages, and the source of the effort to establish the baseline shifts between the project developer (project-specific) and the protocol developer (performance standard). From a program point of view, use of performance standard baselines appears to be increasing. However, the lack of available sector or region-level data quite

9. The Technical Seed Document (single or multiple documents) is the foundation document supporting GHG quantification and approaches in an eventual protocol.

often limits use of this approach, and project-specific baselines must be used.

In scoping the applicable baseline condition, functional equivalence of project and baseline conditions is required, that is, baseline and post-project emissions must be compared with a common base unit, similar to a functional unit in LCA analysis, to calculate meaningful and real emission reductions. Generally, comparisons of baseline and project emissions will be intensity-based. Depending on the protocol and project type, emissions are calculated and reported on a per bushel of crop, per kg of beef or milk, or per unit of energy produced to ensure comparable results.

Quantification

Because agriculture has many mitigation activities with generally small reductions per acre or per unit of production, many programs have developed protocols that rely on standardized quantification approaches to project and baseline calculations. Measurement, monitoring, and verification procedures in these protocols require site-specific data and inputs for some parameters (e.g., acres under a particular management regime, soil disturbance/residue levels to determine tillage frequency, and type of animal feed ration), but standardized quantification approaches, typically based on empirical (e.g., IPCC) or process-based (e.g., DAYCENT or DNDC) models, can be used to estimate emissions under baseline and post-project conditions. These approaches can achieve acceptable levels of quantification accuracy and environmental integrity while reducing project implementation costs (Olander et al. 2011).

Given that GHG emissions from agriculture in most cases are highly variable across landscapes, over time (both interannually and intra-annually), and even within a single field, no single measurement technique is deemed sufficient as a comprehensive GHG measurement or monitoring system applicable in a protocol. Typically, a combination of measurement and modeling is used to estimate GHG emissions in agriculture. Each approach and technology has unique constraints related to costs and sampling design requirements and, hence, each has its level of uncertainty.

As a result, scientists must use a variety of techniques across a range of scales to cross-check the data from any one method and thereby overcome that method's limitations. They can integrate and overlay data with models, either empirical or process based. Then they use the models to compile data from diverse measurements to scale estimates of GHG emissions and reductions from measurement sites to fields, entire farms, or even whole regions. To manage variability and uncertainty, most scientists combine modeling approaches with field data. Some researchers apply models with a Tier 2 level of quantification. These models are well-calibrated with long-term experimental site data measurements unique to the local conditions and are then applied at larger scales to derive standardized quantification approaches for both baseline and project conditions. Others apply Tier 3 modeling with highly specific on-site data and measurement inputs (Olander et al. 2011).

For regional applications, models require input data on several environmental factors, such as agroclimatic data, soil conditions (e.g., type, organic matter, texture), and topography. They also require inputs on farm management activities and site and regional variance in those activities over time. In the United States, remote sensing and national resource inventory datasets provide high-quality spatial data on climate, soil conditions, and topography. GIS-based information systems are also improving the organization and collection of activity data for use in models (Olander et al. 2011; Denef et al. 2011). However, some challenges remain relative to comprehensive coverage of land use and management activity data at the regional level. Protocol developers need to invest in gathering data to support the required performance standard baseline approaches and method uncertainty estimates. The level of uncertainty associated with a particular measurement or estimation technology must be stated in the protocol development process to adjust the volume of credits that should be awarded on the basis of use of that quantification method. Because methane and nitrous oxide are the major gases emitted in the livestock sector, and measurement for livestock is difficult and costly, modeling will likely continue to be the primary method to estimate emissions in the short to medium term.

Additionality

Project-based protocols require that carbon offsets be generated from projects that provide "additional" emissions reductions, that is, not merely emissions reductions from business-as-usual (BAU) activities or those that would have occurred absent the incentive provided by carbon markets. So-called additionality is a fundamental policy requirement of all carbon markets. Currently, the additionality rules for projects can vary greatly across carbon markets and protocols. Most existing and proposed regulatory systems apply some programmatic-level criteria (e.g., project activities must start after a certain date or go beyond regulatory requirements). In addition, the regulatory requirements may establish offset project types. Typically, carbon protocols use other tests to assess additionality, which is a subjective determination of what would have happened had the project not been implemented. These tests include

- ***Common practice tests*** to determine that similar activities in a specific region or at a sector level are not occurring
- ***Barriers tests*** to demonstrate that technological challenges, third-party investment needs, and other barriers to implementation of the project are overcome through generation of emissions reduction credits
- ***Financial tests*** to ensure that the project does not represent the least-cost scenario
- ***Technology benchmarks*** to assess whether a specific technology is being used or to specify that only certain technologies are beyond BAU, e.g., second-generation biofuel production or use of anaerobic digesters
- **Emissions performance thresholds** to set an emissions rate threshold that a project must beat to generate an offset

For agricultural projects, the application of additionality is an evolving concept. Janzen et al. (2011) proposed a set of additionality principles adapted to the particular circumstances of agricultural protocols and projects. They asserted that aggregators or project developers are the agents of change who respond to market interventions. They further asserted that technical agriculture experts and practitioners need to be included in the design of tests to assess how these agents overcome barriers to catalyze practice change and outcomes at meaningful scales. Operationalizing these tests in the multi-faceted context of agriculture is likely best achieved through standardized approaches.

Janzen et al. (2011) make three major recommendations for assessing additionality in agricultural projects/protocols:

- To mobilize emissions reductions in agriculture, given the small tonnage across many dispersed sources, the infrastructure and service capacity generated by offset markets can serve as a catalyst for innovation to overcome barriers to adoption of new practices.
- Farmers, technical agricultural experts, and other experts should make additionality determinations.
- Additionality tests for the agricultural sector should be standardized (embedded in protocols, rather than assessed on a project-by-project basis) and include "positive lists," standardized barriers tests, performance benchmarks, and so on.

Depending on the program or registry, projects must prove they are additional in one of two ways: project-specific tests or performance-standard tests. The former involves assessing whether the project is additional on a case-by-case basis. Under this approach, a project baseline scenario is identified, and any emissions reductions beyond the baseline are considered additional. Standardized tests typically have already been performed by the protocol developer or program manager and are set out in the protocol. Some programs and registries suggest use of standardized additionality approaches and performance standard baseline determinations. But because datasets do not yet support development of these approaches and determinations for all agricultural and livestock GHG mitigation strategies meriting consideration, use of project-specific tests are allowed.

Differences in additionality testing can identify the reasons that some projects and protocols are allowed for use in some systems but not in others.

Leakage

In all voluntary and regulatory markets, projects/protocols must address leakage. Leakage is defined as any change in GHG emissions that occurs outside of a project's boundary (but typically within the same country) that is measurable and attributable to the project's activities (see Jenkins, Olander, and Murray 2009). It can be positive (enhancing mitigation) or negative (offsetting some mitigation benefits). For example, improving pasture management can increase productivity, decreasing the acreage of needed pasture land. Thus, land elsewhere can be taken out of production and managed for enhanced C sequestration, increasing overall mitigation.

Three mandated requirements of application of the ISO 14064:2 standard framework to protocol development are conservatism in baseline estimates, use of a streamlined LCA for SSRs in the baseline and the project, and functional equivalence in comparison of the two. These requirements minimize leakage risk. Systematic assessment of GHG sources and sinks, controlled upstream and downstream from the project site, reduces the chance that any major positive or negative leakage effects will be overlooked. Selection of an appropriate unit of functional equivalence (i.e., per bushel of crop or per kg of beef produced) can take into account any such effects.

In addition, programs can stipulate how to calculate leakage and mitigate leakage risk (or impact). There are a number of analytical approaches to estimating leakage (Olander et al. 2011). In the case of negative leakage, a project may have to surrender a certain percentage of carbon credits to a buffer pool. In general, GHG programs may recognize, but not allow crediting for, positive leakage due to the conservativeness principle and the difficulty of establishing a clear causal link between increased production in the project area and decreased emissions elsewhere.

Validation and verification

Although associated with increased project cost and administrative complexity, a validation process provides an added level of certainty regarding a project's offset eligibility by ensuring that the quantification process in the protocol is correctly applied. In a verification process, an objective third party examines a claim of a reduction in GHGs arising from an offset project and provides an opinion about that claim. The objective of the verification process is to ensure that the GHG assertion is, in the view of the verifier, free of material misstatements and meets a reasonable level of assurance (both of which are defined by the GHG program). The more general objective is to provide confidence that the offset credits awarded in the voluntary or regulatory program are credible and of sufficient quality.

Validation occurs before the project begins (ex ante) and focuses on whether

- appropriate baseline and project conditions are used;
- calculations of potential offsets are correct; and
- protocol eligibility, additionality, leakage, and other conditions have been met.

Validation occurs once per crediting period and requires technical expertise in the project area.

Verification occurs once reductions have been generated (ex post) and focuses on

- correctness of calculations of actual offsets;
- data integrity and consistency with the offset project plan and quantification protocol;
- data completeness, accuracy, and conformance with verification criteria.

Verification occurs at the end of each reporting period.

A key consideration when developing protocols for agriculture and livestock systems is whether the emissions reduction activity and the calculated emissions reductions can be supported with farm and ranch activity data that can reasonably be expected to be available. The protocol must reflect two data considerations: (1) whether the right kinds of farm and ranch activity data can be collected to calculate GHG emissions and (2) whether the farms and ranches have record-keeping systems that can provide appropriate documentation to a third-party validation/verification body to prove the GHG activity data are correct.

GHG calculations can be supported by feeding/ration management software and data management systems for production performance (milk reporting systems, animal weight gains, feedlot close-out data systems, nutrient/manure management systems software). In livestock production, such software and systems are commonly used, and supply contract management is common in meat (poultry and pork) and milk production, so standardization and harmonization of data management systems is occurring. The exception to this trend is in beef production in confined feeding situations, though data availability and uniformity is also increasing in that sector.

Examples of farm records and documents that would support the data in the databases used to calculate GHG reductions include financial records for transactions (feed purchases, animal imports, professional services, milk sales, meat sales, shipping manifests, animal ID tags). Programs and registries increasingly are requiring sign-off from independent service providers and advisors. They deem data from independent sources (crop or animal insurance, custom manure applicators) superior to farm data.

Guidance on verification and validation requirements is set out in programmatic guidance documents for each system. Often, accompanying project plan and project report templates are provided to facilitate consistent validation and verification procedures. Most systems will cite ISO 14064:3 and 14065, or similar accounting standards. These standards set out the frameworks for verification and validation; the criteria against which projects are verified are the programmatic rules, specific quantification protocols, and individual project plans. Organizations like the American National Standards Institute (ANSI) and the UNFCCC have accreditation programs to ensure verifiers and validators are accredited to a common set of principles, standards, and practices within their area of competency. However, delays in implementation of regulatory programs have slowed capacity building in this service.

As noted, the quantification protocols based on the ISO 14064:2 framework contain multiple methods or modules, including detailed information on measurement approaches, baseline conditions, project data monitoring and quantification procedures, reporting, and data management specifications. This information greatly aids verification of GHG emissions reductions and removals for particular project types. Accurate data collection, compilation, and retention are fundamental to verifying an offset project's emissions reductions. Errors in data collection and handling can result in material errors in the reductions being claimed, possibly resulting in financial liability for the project developer. Ensuring that sufficient controls are in place provides credibility to the overall project and associated emissions reductions.

Across voluntary and regulatory carbon markets, third-party verification is a fundamental requirement for carbon offset projects. However, project approval or validation steps vary with each program (e.g., CDM, ACR, and CAR require a project design document to be submitted for approval). Some programs (e.g., ACR and CAR) allow validation and verification to be conducted simultaneously and by the same entity, reducing validation and verification costs.

Protocol development

Generally speaking, protocols can be developed by a program or offered for approval by prospective protocol developers, either private-sector, non-government organizations or public-sector entities. Regardless, a number of features are common to some of the more robust processes: expert and market engagement, defensible scientific and technical methodologies and best practice guidance, a rigorous peer review process, documented transparency in development stages, and final decision making on the part of the regulator or program manager. A clearly defined, transparent development, approval, and revision process ensures that carbon offsets stand up to scrutiny and are realized in a cost-effective and timely manner.

Protocol development unites project-specific scientific knowledge with GHG quantification expertise and familiarity with the programmatic requirements of a particular system. It must occur within a framework (most broadly, ISO 14064:2) and fit within applicable policy and market regimes. These requirements raise consideration of the geographic distribution of projects, the time required for protocol development and approval, and the cost of protocol development. This cost must be weighed against the relevant benefit from the resultant offset value.

If protocol accounting approaches are consolidated, offset amounts under similar protocols might be comparable, once policy and market overlays are taken into account. An in-depth rationale should be included in a protocol to address any differences in treatment across systems.

Table 22. Comparison of LCA and project-based accounting, with an emphasis on GHG emissions, in U.S. livestock production

Comparative aspect	LCA accounting	Project-based accounting
Standards	ISO 14040, 14044 ISO 14067:DIS (Draft C footprint standard); GHG Protocol Product LCA Accounting and Reporting Standard; WRI GHG Protocol Corporate Accounting and Reporting Standard; PAS 2050	ISO 14064:2 and ISO 14064:3 (verification standard) and WRI GHG Protocol Project-Based Accounting Standard
Purpose(s)	Benchmarking for future comparisons of continuous improvements in environmental performance Informing/hotspot identification for decision-makers in industry, government, or NGOs; strategic planning, priority setting, product or process design or redesign; mitigation strategies Comparison of agricultural practices and feed/manure management choices Marketing: eco-label, making an environmental claim, differentiating the products in traditional marketplaces on the basis of carbon intensity within a subsector of livestock production	Access carbon markets established by programs and registries by generating carbon offsets Used as a process standard to develop project-specific GHG quantification protocols or project design documents (depending on the program)
Principles	TBD	Relevance, completeness, consistency, accuracy, conservativeness, transparency
Accounting features		
Scope and boundary	Depending on study goal, user chooses whether to include primary (on-farm), secondary (upstream/downstream), or tertiary input and output inventories. For livestock production, the project boundary typically is cradle to farm gate or cradle to grave. "Cradle" most often includes crop input manufacturing but rarely fuel extraction and processing/machinery manufacturing (14067 :DIS is for GHGs only). The decision to include indirect land use change, and assumptions about why the land use change is occurring (i.e., for livestock grazing, biofuels, the crop, etc.), is controversial, and including it, with or without co- or by-product allocation, can cause wide fluctuations in LCA results.	GHGs only Defined first by conducting a streamlined LCA of controlled (on-site), related (upstream/downstream), and affected (market/activity shifting) SSRs, which is analogous to primary, secondary, and tertiary scope LCIs of the 14040 approach. After comparison of the baseline and project's streamlined LCAs, the project boundary emerges, and relevant sources/sinks with associated inputs/outputs and market shifting impacts/sources are identified for quantification. Criteria of significance and conservativeness are applied to allow project developers to exclude some SSRs from accounting.
Life-cycle impact categories	The user chooses environmental impact categories, which often include GHGs, water use, nutrient impacts (including other gases), land use/environmental footprint. Biodiversity is more common a category in cropping systems LCAs. 14067:DIS is GHG specific. In North America, several livestock studies focus on GHGs only to determine the C footprint of meat or milk production.	Based on material and energy flows into and outside of the project and baseline condition (see above) and quantifies GHGs only. Can exclude some categories from quantification on the basis of a relative comparison of baseline conditions and post-project changes, with proper justification (i.e., if the levels of GHG emissions from the source are less than or the same as in the baseline)

Comparative aspect	LCA accounting	Project-based accounting
Functional unit	Varies between kg live weight, kg carcass weight, kg of retail cuts, kg of milk, kg of energy-corrected milk (adjusting for fat and protein content); MJ of energy, kg of protein. No consistency; comparison of studies problematic.	Unit of comparison of baseline and post-project conditions is called "functional equivalence." The metric must be based on one level of service or product, necessitating an intensity-based metric for comparison.
Allocation (leakage)	Because several co- or by-products exist in agricultural production and the multifunctional input/output streams of livestock and crop production affect other production systems, allocation is necessary. ISO identifies a hierarchy of allocation: system expansion; physical relationships/causality; composition and economic value. Most often, livestock LCAs use economic or mass because of the complex data requirements and availability of system expansion.	Allocation is not explicit, but leakage is a related concept. The streamlined LCA included in SSR analysis does identify input streams that may be byproducts of other production systems (e.g., corn DDGS). In this case, the user must assess the emissions contribution of DDGS entering the system from this feedstock.
Uncertainty	Not often performed or required in agricultural LCAs; often sensitivity analysis is performed to assess impact of key parameters.	Key requirement of the standard and of programs/registries. Current practice is to understand structural and input uncertainty for models. Uncertainty discounts are applied on the basis of the conservativeness principle.
Additionality	Not typically a criterion; benchmarking and assessing continuous improvements over time. No exclusion of BAU activities.	Key policy requirement of programs/registries; defined in different ways and to different degrees
Life-cycle evaluation techniques	Explicit requirement. Quantification of all life-cycle stages is set out in the chosen LCI; all LCI impact categories identified by the user must be quantified within the scope and boundary set out by the goal phase. However, when the goal is to compare systems, users will exclude sources because they are the same; the goal in this case determines the magnitude of the impact.	Streamlined LCA identification between baseline and project SSRs is based on material and energy or input/output flows; only relevant SSRs need to be quantified (typically based on 5% materiality threshold to overall carbon reduction).
Data quality and assessment	Transparency is important to interpret the usefulness and legitimacy of the results. Should comply with the standard and include a description of the quality of the data (data inputs for primary and secondary sources) so the user can understand the reliability of the study and interpret the results. Increasingly, the LCA community and stakeholders are calling for and moving toward open source code, inventory databases, and emissions factors unique to agricultural production systems.	ISO 14064:2 and programs/registries identify the need for strong data management systems with QA/QC checks and controls to reduce the likelihood of errors. Data controls and QA/QC procedures are needed to ensure data are complete, accurate, valid, and not subject to corruption. Restricted access to data is typically required to address the security of the data management system.
Permanence	Accounting of reversals and implementation of replacement mechanisms (discounts, buffer reserves, direct replacements, temporary carbon) are not required.	A main policy criterion for project types with a risk of reversal (i.e., carbon sequestration projects). Not applicable for project types avoiding CO_2, N_2O, or CH_4 emissions when these emissions avoidances cannot subsequently be reversed.
MMRV requirements		
Quantification methodology and procedures	Because most LCA livestock studies are cradle to farm gate and focused on a limited amount of production emissions sources upstream (feed production inputs but not machinery, medicine manufacturing, fuel extraction, and processing), on-farm sources of CH_4, N_2O, and fossil CO_2 tend to be considered. Most studies applied either IPCC approaches or simulation modeling for N_2O and CH_4 emissions from enteric emissions and manure, with an emphasis on IPCC. Most often, Tier 2 modeling is used for ruminants and Tier 1 for monogastrics. For on-farm fuel/energy use, a variety of sources are used for quantification. Very few LCAs quantify soil carbon sequestration. Ym factors for enteric CH_4 emissions varied, as did the GWP for the various gases. Vergé et al. (2011) make the case for considering production systems as a whole, including broad ecosystem interactions, corresponding to a full system expansion, because ecosystems are spatially defined, and impacts are related to spatial and temporal variations.	To meet the "real and demonstrable" policy criterion of carbon offset programs and registries, the completeness principle of ISO 14064:2 must be met, meaning all GHGs affected by the project must be considered and accounted for on a net impact basis. Livestock protocols in this space are few and mostly rely on IPCC Tier 2 approaches (accounting for the animal category, diet characteristics that drive GHG emissions, dry matter intake, nitrogen and volatile solids excretion). The interaction with livestock production on a net GHG accounting basis reflects relevant upstream and downstream impacts and brings them into quantification procedures. The available protocols could benefit from well-calibrated process-based models; most focus on confined feeding configurations because of data management and availability.
Monitoring of data sources	Most practitioners and policy makers recognize that LCI inventory input datasets should be based on regional, primary data collection from the sector or commodity being modeled. However, most studies use existing data and not farm-based data for primary sources. Secondary sources tend to be from well-published and documented sources.	Project-based accounting typically uses actual project data gathered on each farm to drive GHG calculations. However, evidence to support the GHG calculations and assertions of GHG reductions or offsets resulting from project-based accounting is a requirement over and above LCA accounting. The purpose drives the difference. Farm-based evidence must be gathered to support GHG calculations.
Reporting	Depends on the user of the information and the goal of the study. If intended to be disclosed to the public for comparability or marketing purposes, reporting can be ISO 14040 compliant only if reviewed by key experts. For eco-labeling, a cradle-to-grave cycle may be required.	Offset programs, whether voluntary or regulatory, have specific reporting requirements, using project report templates. Offset project plans, third-party validation reports, project reports, and GHG assertions as well as third-party verification reports are typically posted on-line for transparency and accountability.
Third-party verification	Depending on the goal, a third-party critical review may take place.	The verifier will assess quantitative and qualitative material discrepancies in GHG quantification and accounting on a project basis. The verification is formalized, typically by program-level verification standards; accreditation programs certify verifiers by competency. A program-level review of actual project verification activities is common.

6. CONCLUSIONS

This report synthesizes the fundamental information necessary for designing programs to report and mitigate enteric and manure methane emissions from livestock in the United States. This information, like that contained in a companion T-AGG report on reducing GHG emissions from cropping systems and rangelands, could contribute to the USDA's effort to incorporate mitigation objectives into agricultural incentive and extension programs. Likewise, this report could promote exploration of agriculturally related GHG reduction opportunities relative to certification programs for agricultural products and corporate sustainability initiatives focused on the supply chain. It also could help the agricultural sector participate in California's developing compliance-based carbon offsets market.

Although a small contributor to overall GHG emissions, livestock management is a significant contributor to total CH_4 emissions in the United States. Given relatively steady U.S. livestock production numbers, emissions reductions are more likely to be achieved through changes in management of existing production systems than by whole-scale shifts in these systems. Opportunities to reduce CH_4 emissions through feed and manure management changes are significant. Each livestock production system is different, and no one management change will work for every operation. However, educational resources identifying a suite of economically sustainable options and the impact of adoption will allow producers to identify those practices that can fit their system.

For beef and dairy cattle, a number of strategies are linked to improving productivity by reducing CH_4 emissions per unit of product. Improved diet and genetics make these animals use feed more efficiently and reach maturity more quickly. Other strategies focus on reducing enteric emissions through feed additives like ionophores and organic acids. Although limited in scale, improved pasture management also holds promise for reducing CH_4 emissions. Other strategies need additional research; most are expected to have moderate to low mitigation potential. For swine, only one feeding strategy, lowering protein, appears to have mitigation potential.

For beef and dairy cattle, the most promising manure management strategies include aeration, compaction, and composting and the shift from liquid to solid systems. The mitigation potential and financial viability of these strategies should be further researched. The most well-studied and high-potential activity is use of a digester for CH_4 capture and flaring and for CH_4 combustion to produce energy. This activity is already incorporated in many carbon offsets programs and renewable energy programs. However, the capital, operation, and maintenance costs of changing manure handling systems can present a significant hurdle. Overcoming it could require external investment and changes in energy sector policy and business models.

For swine, many manure management strategies appear promising in terms of GHG mitigation potential. These strategies include increased manure removal frequency, solid-liquid separation, optimized bedding materials for dry manure management, covered manure storage, and composting. The mitigation potential and financial viability of these strategies should be further researched. Covering manure may have only moderate costs, and composting could generate money through production of a value-added fertilizer for off-site marketing. Anaerobic digestion, although costly, also has high mitigation potential and is ready for use in programs similar to those developed for dairy cattle manure digesters.

Emissions reductions from manure CH_4 capture systems can be measured directly. Otherwise, various modeling approaches are likely to be used to quantify emissions reductions from livestock systems. Many models are being tested and used for national inventory quantification. A range of farm-scale accounting tools for carbon offsets programs as well as corporate supply chain life-cycle assessment (LCA) tools are in development. These models and system-level tools are needed to ensure that management changes do not inadvertently increase one greenhouse while reducing another.

Accounting approaches for tracking GHG emissions from livestock are also in development. Carbon offset protocols are designed to track emissions changes resulting from a specific project (or management change). LCAs, on the other hand, are used to assess changes in a production system or supply chain for a product; they indicate emissions hot spots and facilitate emissions comparisons across supply chains, products, and time. As yet, LCA methods have little consistency from one application to another.

In summary, livestock management has a few well-researched and ready-for-action opportunities for making measurable and substantial methane reductions. The more established opportunities, like anaerobic digesters, tend to be costly. The mitigation potential and financial viability of many other potentially significant management opportunities should be further researched.

REFERENCES

Aarnink, A.J.A., P. Hoeksma, and E.N.J. van Ouwerkerk. 1993. Factors affecting ammonium concentration in slurry from fattening pigs. In *Nitrogen Flow in Pig Production and Environmental Consequences,* ed. M.W.A Verstegen, L.A. den Hartog, G.J.M. van Kempen, and J.H.M. Metz. The Netherlands: Wageningen Academic Publishers.

Aarnink, A.J.A., and M.W.W. Verstegen. 2007. Nutrition, key factor to reduce environmental load from pig production. *Livestock Science* 109:194–203.

Alemu, A.W., J. Dijkstra, A. Bannink, J. France, and E. Kebreab. 2011a. Rumen stoichiometric models and their contribution and challenges in predicting enteric methane production. *Animal Feed Science Technology* 166–167:761–778.

Alemu, A.W., K. Ominski, and E. Kebreab. 2011b. Trends of enteric methane emissions from Manitoba beef cattle. *Canadian Journal Animal Science* 91:305–321.

Amlinger. F., S. Peyr, and C. Cuhls. 2008. Greenhouse gas emissions from composting and mechanical biological treatment. *Waste Management and Research* 26:47–60.

Amon, B., Kryvoruchko, V., Amon, T., Beline, F., Petersen, S., 2004. Quantitative effects of storage conditions on GHG emissions from cattle slurry, and N_2O and CH_4 turnover inside natural surface crusts. Final Reports of the MIDAIR project "Greenhouse Gas Mitigation for Organic and Conventional Dairy Production," Deliverable No. 5.5. Institute for Energy and Environment, Leipzig, Germany.

Amon, B., V. Kryvoruchko, T. Amon, and S. Zechmeister-Boltenstern. 2006. Methane, nitrous oxide, and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. *Agriculture, Ecosystems & Environment* 112:153–162.

Amon, B., V. Kryvoruchko, M. Frohlich, T. Amon, A. Pollinger, I. Mosenbacher, and A. Hausleitner. 2007. Ammonia and greenhouse gas emissions from a straw flow system for fattening pigs: Housing and manure storage. *Livestock Science* 112:199–207.

Archibeque, S.L., H.C. Freetly, N.A. Cole, and C.L. Ferrell. 2007. The influence of oscillating dietary protein concentrations on finishing cattle. II. Nutrient retention and ammonia emissions. *Journal of Animal Science* 85:1496–1503.

Atakora, J.K.A., S. Moehn, and R.O. Ball. 2004. Very low protein diet for finishing pigs maintains performance and reduces greenhouse gas production. Paper presented at the annual meeting for the Society of Animal Science, Edmonton, Alberta, July 21–23.

Atakora, J.K.A., S. Moehn, and R.O. Ball. 2003. Low protein diets maintain performance and reduce greenhouse gas production in finisher pigs. *Advances in Pork Production* 14: Abstract #17.

Axelsson, J. 1949. The amount of produced methane energy in the European metabolic experiments with adult cattle. *Annual Review of Agricultural Colleges in Sweden* 16: 404–419.

Baldwin, R.L. 1995. *Modeling Ruminant Digestion and Metabolism.* London: Chapman & Hall.

Ball, R.O., and S. Möhn. 2003. Feeding strategies to reduce greenhouse gas emissions from pigs. *Advances in Pork Production* 14:301.

Bannink A., J. Kogut, J. Dijkstra, J. France, E. Kebreab, A.M. Van Vuuren, and S. Tamminga. 2006. Estimation of the stoichiometry of volatile fatty acid production in the rumen of lactating cows. *Journal of Theoretical Biology* 238:36–51.

Bannink, A., M.W. van Schijdel, and J. Dijkstra. 2011. A model of enteric fermentation in dairy cows to estimate methane emission for the Dutch National Inventory Report using the IPCC Tier 3 approach. *Animal Feed Science Technology* 166–167:603–618.

Barker, J.C. 1996. Swine production system management: Open dirt or pasture lots. Raleigh: North Carolina Cooperative Extension Service. http://www.bae.ncsu.edu/programs/extension/publicat/wqwm/ebae179_93.html.

Basarab, J., V. Baron, Ó. López-Campos, J. Aalhus, K. Haugen-Kozyra, and E. Okine. 2012. Greenhouse Gas Emissions from Calf- and Yearling-Fed Beef Production Systems, With and Without the Use of Growth Promotants. *Animals*. 2:195–220.

Bates, J. 2001. Economic evaluation of emission reductions of nitrous oxides and methane in agriculture in the EU. Abingdon, UK: AEA Technology Environment. http://ec.europa.eu/environment/enveco/climate_change/pdf/agriculture.pdf.

Bayat. A., and K.J. Shingfield. 2012. Overview of nutritional strategies to lower enteric methane emissions in ruminants. Maataloustieteen Päivät. http://www.smts.fi/Kotielaintuotanto/Bayat_Overview%20of.pdf.

Beauchemin, K.A., H.H. Janzen, S.M. Little, T.A. McAllister, S.M. McGinn. 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agricultural Systems* 103 371–379.

Beauchemin, K.A., H.H. Janzen, S.M. Little, T.A. McAllister, S.M. McGinn. 2011. Mitigation of greenhouse gas emissions from beef production in western Canada – Evaluation using farm-based life cycle assessment. Animal Feed Science and Technology 166– 167 (2011) 663– 677.

Beauchemin K.A., S.M. McGinn, C. Benchaar, and L. Holtshausen. 2009. Crushed sunflower, flax, or canola seeds in lactating dairy cow diets: Effects on methane production, rumen fermentation, and milk production. *Journal of Dairy Science* 92:2118–2127.

Beauchemin, K.A., M. Kreuzer, F. O'Mara, and T.A. McAllister. 2008. Nutritional management for enteric methane abatement: A review. *Australian Journal of Experimental Agriculture* 48:21–27.

Benchaar, C., C. Pomar, and J. Chiquette. 2001. Evaluation of dietary strategies to reduce methane production in ruminants: A modeling approach. *Journal of Animal Science* 81:563–574.

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* 76:617–627.

Bennett, E.M and P. Balvanera. 2007. The future of production systems in a globalized world. *Frontiers in Ecology and the Environment* 5: 191–198.

Benz, D.A., and D.E. Johnson. 1982. The effect of monensin on energy partitioning by forage fed steers. In *Proceedings, Western Section, American Society of Animal Science* 33:60.

Berg, W.E. 2003. Reducing ammonia emissions by combining covering and acidifying liquid manure. In *Air Pollution from Agricultural Operations III, Conference Proceedings*, ed. Robert Burns, 174–182. Raleigh, North Carolina: American Society of Agricultural and Biological Engineers.

Berg, W., and I. Pazsiczki. 2003. Reducing emissions by combining slurry covering and acidifying. In *Proceedings of the International Symposium on Gaseous and Odour Emissions from Animal Production Facilities*, 460–468. Horsens, Denmark: International Symposium on Gaseous and Odour Emissions from Animal Production Facilities.

Bhatti, J.S., R. Lal, M.J. Apps, and M.A. Price, eds. 2005. *Climate Change and Managed Ecosystems*. Boca Raton: CRC Press.

Blanes-Vidal, V., M.N. Hansen, S. Pedersen, and H.B. Rom. 2008. Emissions of ammonia, methane, and nitrous oxide from pig houses and slurry: Effects of rooting material, animal activity, and ventilation flow. *Agriculture, Ecosystems and Environment* 124:237–244.

Blaxter, K.L., and J.L. Clapperton. 1965. Prediction of the amount of methane produced by ruminants. *British Journal of Nutrition* 19: 511–522.

Boadi, D.A., K.M. Wittenberg, and A.D. Kennedy. 2002. Validation of the sulphur hexafluoride (SF_6) tracer gas technique for measurement of methane and carbon dioxide production by cattle. *Canadian Journal of Animal Science* 82:125–131.

Boldrin, A., J.K. Andersen, J. Møller, T.H. Christensen, and E. Favoino. 2009. Composting and compost utilization: Accounting of greenhouse gases and global warming contributions. *Waste Management & Research* 27: 800.

Bridges T.C., L.W. Turner, G.L. Cromwell, and J.L. Pierce. 1995. Modeling the effects of diet formulation on nitrogen and phosphorus excretion in swine waste. *Applied Engineering in Agriculture* 11:731–738.

Brown, S., C. Kruger, and S. Subler. 2008. Greenhouse gas balance for composting operations. *Journal of Environmental Quality* 37:1396–1410.

Brumm, M.C., J.D. Harmon, M.S. Honeyman, J.B. Kliebenstein, and J.M. Zulovich. 1999. Hoop structures for gestating swine. Staff General Research Paper 1474, Department of Economics, Iowa State University.

Cabaraux, J.F., F.X. Philippe, M. Laitat, B. Canart, M. Vandenheede, and B. Nicks. 2009. Gaseous emissions from weaned pigs raised on different floor systems. *Agriculture, Ecosystems, and Environment* 130:86–92.

Cammell, S.B., D.J. Thomson, D.E. Beever, M.J. Haines, M.S. Dhanoa, and M.C. Spooner. 1986. The efficiency of energy utilisation in growing cattle consuming fresh perennial ryegrass (*Lolium perenne* cv. Melle) or white clover (*Trifolium repens* cv. Blanca). *British Journal of Nutrition* 55:669–680.

Canh, T.T., A.J.A. Aarnink, J.B. Schutte, A.L. Sutton, D.J. Langhout, and J.W. Schrama. 1998. Dietary protein affects nitrogen excretion and ammonia emission from slurry of growing finishing pigs. *Livestock Production Science* 56:181–191.

Capper, J.L. 2011. The environmental impact of beef production in the United States: 1977 compared with 2007. *Journal of Animal Science* 89:4249–4261.

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* 87:2160–2167.

CAST (Council for Agricultural Science and Technology). 2011a. *Air issues associated with animal agriculture: A North American perspective*. CAST Issue Paper 47. Ames Iowa: The Council.

CAST. 2011b. *Carbon sequestration and greenhouse gas fluxes in agriculture: Challenges and opportunities*. Task Force Report No. 142. Ames, Iowa: The Council.

CEART, Inc. 2011. The stages of pork production. http://www.midbuchanan.k12.mo.us/organizations/ffa/AGI%20Material/Swine%20Production/Assignments/The%20Stages%20of%20Pork%20Production%20(MyCAERT).pdf.

Christensen, K., and G. Thorbek. 1987. Methane excretion in the growing pig. *British Journal of Nutrition* 57:355–361.

Clark, H., C.A.M. de Klein, and P. Newton. 2001. Potential management practices and technologies to reduce nitrous oxide, methane, and carbon dioxide emissions from New Zealand agriculture. Ministry of Agriculture and Forestry, Wellington, New Zealand.

Clark, O.G., S. Moehn, I. Edeogu, and J. Leonard. 2005. Manipulation of dietary protein and nonstarch polysaccharide to control swine manure emissions. *Journal of Environmental Quality* 34:1461–1466.

Clemens, J., and H.J. Ahlgrimm. 2001. Greenhouse gases from animal husbandry: Mitigation options. *Nutrient Cycling in Agroecosystems* 60:287–300.

Clough, T.J., R.R. Sherlock, K.C. Cameron, and S.F. Ledgard. 1996. Fate of urine nitrogen on mineral and peat soils in New Zealand. *Fertilizer Research* 178:141–152.

Cooprider K.L., F.M. Mitloehner, T.R. Famula, E. Kebreab, Y. Zhao, A.L. Van Eenennaam. 2011. Feedlot efficiency implications on greenhouse gas emissions and sustainability. *Journal of Animal Science* 89:2643–2656.

Corré, W.J. 2002. Agricultural land use and emissions of methane and nitrous oxide in Europe. *Plant Research International*. Wageningen, The Netherlands.

Costa, A., and M. Guarino. 2008. Photocatalytic TIO2 treatment effects on ammonia, greenhouse gases, dust concentration and emission in a weaning room. In *Agricultural and Biosystems Engineering for a Sustainable World*. Presented at the International Conference on Agricultural Engineering and Industry Exhibition, Hersonissos, Crete, Greece, June 23–25.

Cottle, D.J., J.V. Nolan, and S.G. Wiedemann. 2011. Ruminant enteric methane mitigation: A review. *Animal Production Science* 51:491–514.

Crutzen, P.J., I. Aselmann, and W. Seiler. 1986. Methane production by domestic animals, wild ruminants, other herbivorous fauna, and humans. *Tellus B* 38B:271–284.

Cullimore, D.R., A. Maule, and N. Mansuy. 1985. Ambient temperature methanogenesis from pig manure waste lagoons: Thermal gradient incubator studies. *Agricultural Wastes* 12:147–157.

Denef, K., S. Archibeque, and K. Paustian, 2011. Greenhouse gas emissions from U.S. agriculture and forestry: A review of emission sources, controlling factors, and mitigation potential. Interim report to USDA under Contract #GS23F8182H.

de Vries, M., and I.J.M. de Boer. 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. *Livestock Science* 128:1–11.

DeRamus, H. Alan, Terry Clement, Dean Giampola, and Peter Dickison. 2003. Beef production efficiency in forages and grazing management systems as monitored by methane emissions. *Journal of Environmental Quality* 32:269–277.

DeSutter, T.M., and J.M. Ham. 2005. Lagoon-biogas emissions and carbon balance estimates of a swine production facility. *Journal of Environmental Quality* 34:198–206.

Dijkstra J., H.D. St. C. Neal, D.E. Beever, and J. France. 1992. Simulation of nutrient digestion, absorption, and outflow in the rumen: Model description. *Journal of Nutrition* 122:2239–2256.

Dinuccio, E, P. Balsari, and W. Berg. 2008. GHG emissions during the storage of rough pig slurry and the fractions obtained by mechanical separation. *Australian Journal of Experimental Agriculture* 48:93–95.

Dittert, K., R. Bol, R. King, D. Chadwick, and D. Hatch. 2001. Use of a novel nitrification inhibitor to reduce nitrous oxide emissions from 15N labelled dairy slurry injected into the soil. *Rapid Communications in Mass Spectrometry* 15:1291–1296.

Dong, H., Z. Zhu, X. Tao, Y. Chen, H. Xin, and J.D. Harmon. 2009. Ammonia, methane, and carbon dioxide concentrations and emissions of a hoop grower-finisher swine barn. Translated by the American Society of Biological Engineers. 52(5):1741–1747.

Dourmard, J.Y., M. Hassouna, P. Robin, N. Guingand, M.C. Meunier-Salaun, and B. Lebret. 2009. Influence of pig rearing system on animal performance and manure composition. *Animal* 3–4:606–616.

Driver, K.S., K.L. Haugen-Kozyra, and R. Janzen. 2010. *Agriculture Sector Greenhouse Gas Practices and Quantification Review*. Agriculture and Carbon Markets: Making Carbon Count Phase 1 Report. Sustainable Food Laboratory. http://sustainablefood.org/projects/ag-carbon-mkts.

Dyer, J.A., X. Vergé, R.L. Desjardins, and D. Worth. 2008. Long term trends in the GHG emissions from the Canadian Dairy industry. *Canadian Journal of Soil Science*. 88: 629–639.

Eagle A.J., L.R. Henry, L.P. Olander, K. Haugen-Kozyra, N. Millar, and G.P. Robertson. 2011. Greenhouse gas mitigation potential of agricultural land management in the United States: A synthesis of the literature. Technical Working Group on Agricultural Greenhouse Gases (T-AGG) Report. http://nicholasinstitute.duke.edu/ecosystem/land/TAGGDLitRev.

Eagle, A.J., L.P. Olander, L.R. Henry, K. Haugen-Kozyra, N. Millar, and G.P. Robertson. 2012. *Greenhouse Gas Mitigation Potential of Agricultural Land Management in the United States: A Synthesis of the Literature, 3rd ed.* Technical Working Group on Agricultural Greenhouse Gases. Nicholas Institute for Environmental Policy Solutions Report.

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. *Livestock Science* 130:47–56.

Ellis J.L., E. Kebreab, N.E. Odongo, B.W. McBride, E.K. Okine, and J. France. 2007. Prediction of methane production from dairy and beef cattle. *Journal of Dairy Science* 90:3456–3467.

Ellis, J.L., A. Bannink, J. France, E. Kebreab, and J. Dijkstra. 2010. Prediction of enteric methane production by dairy cows in whole farm models. *Global Change Biology* 16:3246–3256.

Executive Order 13514. Federal Leadership in Environmental, Energy, and Economic Performance. *Federal Register* 74, no. 194 (October 2009): 52117–52127.

FAO (Food and Agriculture Organization). 2009. *FAOSTAT Statistical Database*. Food and Agriculture Organization of the United Nations. http://apps.fao.org.

———. 2010. Greenhouse gas emissions from the dairy sector: A life-cycle assessment. Rome, Italy: Food and Agriculture Organization.

Felssa, H.P., P. Dorsch, F. Beese, H. Konig, and A.F. Bouwman. 1996. Influence of cattle excrements on nitrous oxide and methane fluxes in pasture land. *Journal of Environmental Quality* 25:1366 Prentice Hall. 1370.

Field, T.G., and R.E. Taylor. 2008. *Scientific Farm Animal Production*, *9th ed.* Columbus, Ohio: Prentice Hall.

Freetly, H.C., and T.L. Brown-Brandl. 2011. Methane emissions from cattle differing in feed intake and feed efficiency fed a high concentrate diet. *Journal of Animal Science* 89: E-Supplement: 33 (abstr.).

Fukumoto, Y., T. Osada, D. Hanajima, and K. Haga. 2003. Patterns and quantities of NH3, N_2O, and CH_4 emissions during swine manure composting without forced aeration-effect of compost pile scale. *Bioresource Technology* 89:109–114.

Gatel, F., and F. Grosjean. 1992. Effect of protein content of the diet on nitrogen excretion by pigs. *Livestock Production Science* 31(1–2):109–120.

Gerber, P., T. Vellinga, C. Opio, and H. Steinfeld. 2011. Productivity gains and greenhouse gas emissions intensity in dairy systems. *Livestock Science* 139(1–2):100–108.

Giger-Reverdin, S., P. Morand-Fehr, and G. Tran. 2003. Literature survey of the influence of dietary fat composition on methane production in dairy cattle. *Livestock Production Science* 82:73–79.

Gill, F.L, R.J. Dewhurst, R.P. Evershed, E. McGeough, P. O'Kiely, R.D. Pancost, and I.D. Bull. 2011. Analysis of archaeal ether lipids in bovine faeces. *Animal Feed Science and Technology* 166–167:87–92.

Gill, F.L., R.J. Dewhurst, J.A.J. Dungait, R.P. Evershed, L. Ives, C. Li, R.D. Pancost, M. Sullivan, S. Bera, and I.D. Bull. 2010. Archaeol: A biomarker for foregut fermentation in modern and ancient herbivorous mammals? *Organic Geochemistry* 41:467–472.

Goel, G., and H.P.S. Makkar. 2011. Methane mitigation from ruminants using tannins and saponins. *Tropical Animal Health Prod.* 44:729–739.

Grainger, C., T. Clarke, S.M. McGinn, M.J. Auldist, K.A. Beauchemin, M.C. Hannah, G.C. Waghorn, H. Clark, and R.J. Eckard. 2007. Methane emissions from dairy cows measured using the sulfur hexafluoride (SF_6) tracer and chamber techniques. *Journal of Dairy Science* 90:2755–2766.

Grainger, C., and K.A. Beauchemin. 2011. Can enteric methane emissions from ruminants be lowered without lowering their production? *Animal Feed Science and Technology* 166–167:308–320.

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* 84:1896–1906.

Guarino, M, A. Costa, and M. Porro. 2008. Photocatalytic TiO2 coating to reduce ammonia and greenhouse gases concentration and emission from animal husbandries. *Bioresource Technology* 99:2650–2658.

Guarino, M., C. Fabbri, M. Brambilla, L.Valli, and P. Navarro. 2006. Evaluation of simplified covering systems to reduce gaseous emissions from livestock manure storage. Translated by the American Society of Biological Engineers 49(3):737–747.

Haeussermann, A., E. Hartung, E. Gallmann, and T. Jungbluth. 2006. Influence of season, ventilation strategy, and slurry removal on methane emissions from pig houses. *Agriculture, Ecosystems & Environment* 112(2–3):115–121.

Hales, K.E., N.A. Cole, and J.C. MacDonald. 2011. 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* 89: E-Supplement: 196 (abstr.).

Hamilton, S.W., E.J. DePeters, J.A. McGarvey, J. Lathrop, and F.M. Mitloehner. 2010. Greenhouse gas, animal performance, and bacterial population structure responses to dietary monensin fed to dairy cows. *Journal of Environmental Quality* 39:106–114.

Hansen, M.N., K. Henriksen, and S.G. Sommer. 2006. Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: Effects of covering. *Atmospheric Environment* 40:4172–4181.

Hao, X., C. Ching, F.J. Larney, and G.R. Travis. 2001. Greenhouse gas emissions during cattle feedlot manure composting. *Journal of Environmental Quality* 30:376–386.

Harper, L.A., R.R. Sharpe, and T.B. Parkin. 2000. Gaseous nitrogen emissions from anaerobic swine lagoons: Ammonia, nitrous oxide, and dinitrogen gas. *Journal of Environmental Quality* 29:1356–1365.

Harper, L.A., R.R. Sharpe, T.B. Parkin, A. De Visscher, O. van Cleemput, and F.M. Byers. 2004. Nitrogen cycling through swine production systems: Ammonia, dinitrogen, and nitrous oxide emissions. *Journal of Environmental Quality* 33:1189–1201.

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* 166–167:227–239.

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: 1392–1401.

Harris, S., and V. Narayanaswamy. 2009. *A Literature Review of Life-cycle Assessment in Agriculture*. Publication 09/029, Project No. PRJ-002940. Rural Industries Research and Development Corporation.

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* 85:1479–1486.

Hellwinckel, C., and J. Phillips. 2012. Land use carbon implications of a reduction in ethanol production and an increase in well-managed pastures. *Carbon Management* 3(1): 27–38.

Hemansen, J.E., and T. Kristensen. 2011. Management options to reduce the carbon footprint of livestock products. *Animal Frontiers* 1(1):33–39.

Hobbs, P.J., B.F. Pain, R.M. Kay, and P.A. Lee. 1996. Reduction of odourous compounds in fresh pig slurry by dietary control of crude protein. *Journal of the Science of Food and Agriculture* 71:508–514.

Holter, J.B., and A.J. Young. 1992. Methane prediction in dry and lactating Holstein cows. *Journal of Dairy Science* 75:2165–2175.

Honeyman, M.S., J. Kliebenstein, and J. Harmon. 2001. Iowa hoop structures used for swine: A survey. ASL-R1780. Swine Research Report AS-646. Ames, IA: Iowa State University Extension.

Husted, S. 1994. Seasonal-variation in methane emission from stored slurry and solid manures. *Journal of Environmental Quality* 23:585–592.

Huther, L., F. Schuchardt, and T. Willke. 1997. Emissions of ammonia and greenhouse gases during storage and composting of animal manures. In *Ammonia and Odour Control from Animal Production Facilities*, EurAgEng, Hrsg., S. 327–334. Presented at the International Symposium, Vinkeloord, The Netherlands, Octoer 6–10.

Innovation Center for U.S. Dairy. 2010. *U.S. Dairy Sustainability Commitment Progress Report.* Rosemont, IL. http://www.usdairy.com/sustainability/Pages/Home.aspx

IPCC (Intergovernmental Panel on Climate Change). 1997. *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, vol. 1–3*. Paris, France: IPCC, United Nations Environment Programme, Organisation for Economic Co-operation and Development, International Energy Agency.

IPCC. 2000. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories, Chapter 4, IPCC National Greenhouse Gas Inventories Programme Technical Support Unit, Kanagawa, Japan.

———. 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories*, ed. by HS Eggleston, L Buendia, K Miwa, T Ngara, K Tanabe. Hayama, Kanagawa, Japan: Prepared by the National Greenhouse Gas Inventories Programme.

International Standard Organization. 2006. ISO 14044. Environmental management—Life cycle assessment. Requirements and guidelines. ISO, Geneva, Switzerland

Jacob, K., A. Atakora, S. Möhn, and R.O. Ball. 2003. Low protein diets maintain performance and reduce greenhouse gas production in finisher pigs. *Advances in Pork Production* 14: Abstract #17.

Janzen, R., T. Baumann, L. Dyer, P. Hardy, and D. Reed. 2011. Additionality in Agricultural Offset Protocols. Presented at the C-AGG meeting, Washington, DC November 2–3. http://www.c-agg.org/resources.html.

Jarvis, S.C, J.R. Wilkins, and B.F. Pain. 1996. Opportunities for reducing the environmental impact of dairy farm management: A systems approach. *Grass Forage Science* 51:21–31.

Jenkins, A., L. Olander, and B. Murray. 2009. Addressing leakage in a greenhouse gas mitigation offsetts program for forestry and agriculture. In *Mitigation Beyond the Cap: A Series of Briefs on Expanding Climate Mitigation Opportunities.* Durham, NC: Nicholas Institute for Environmental Policy Solutions, Duke University.

Jensen, B.B., and H. Jørgensen. 1994. Effect of dietary fiber on microbial activity and microbial gas production in various regions of the gastrointestinal tract of pigs. *Applied and Environmental Microbiology* 60:1897–1904.

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* 35:1064–1068.

Johnson, K.A., and D.E. Johnson. 1995. Methane emissions from cattle. *Journal of Animal Science* 73:2483–2492.

Johnson, K.A., M.T. Huyler, H.H. Westberg, B.K. Lamb, and P. Zimmerman. 1994a. Measurement of methane emissions from ruminant livestock using a SF_6 tracer technique. *Environmental Science Technology* 28:359–362.

Johnson, K.A., M.T. Huyler, H.H. Westberg, B.K. Lamb, and P. Zimmerman. 1994b Measurement of methane emissions from ruminant livestock using a sulfur hexafluoride tracer technique. In *Energy Metabolism of Farm Animals,* ed. by J.F. Aquilera, 335–338. EAAP Publication No. 76. CSIC Publishing Service. Granada, Spain.

Jones, F.M., F.A. Phillips, T. Naylor, and N.B. Mercer. 2011. Methane emission from grazing Angus cows selected for divergent residual feed intake. *Animal Feed Science and Technology* 166–167:302–307.

Jordan, E., D. Kenny, M. Hawkins, R. Malone, D.K. Lovett, and F.P. O'Mara. 2006a. Effect of refined soy oil or whole soybeans on methane output, intake, and performance of young bulls. *Journal of Animal Science* 84:2418–2425.

Jordan, E., D.K. Lovett, F.J. Monahan, and F.P. O'Mara. 2006b. Effect of refined coconut oil or copra meal on methane output, intake and performance of beef heifers. *Journal of Animal Science* 84:162–170.

Jordan, E., D.K. Lovett, M. Hawkins, J. Callan, and F.P. O'Mara. 2006c. The effect of varying levels of coconut oil on intake, digestibility and methane output from continental cross beef heifers. *Animal Science* 82:859–865.

Jørgensen, H. 2007. Methane emission by growing pigs and adult sows as influenced by fermentation. *Livestock Science* 109:216–219.

Jouany, J.P., and D.P. Morgavi. 2007. Use of "natural" products as an alternatives to antibiotic feed additives in ruminant production. *Animal* 1:1443–1466.

Judd, M.J., F.M. Kelliher, M.J. Ulyatt, K.R. Lassey, K.R. Tate, I.D. Shelton, M.J. Harvey, and C.F. Walker. 1999. Net methane emissions from grazing sheep. *Global Change Biology* 5:647–657.

Kaharabata, S.K., P.H. Schuepp, and R.L. Desjardins. 1998. Methane emissions from above ground open manure slurry tanks. *Global Biogeochemical Cycles* 12:545–554.

Kai, P., B. Kaspers, and T. van Kempen. 2006. Modeling source gaseous emissions in a pig house with recharge pit. *Transactions of the American Society of Agricultural and Biological Engineers.* 49(5):1479–1485.

Kebreab E., J.A.N. Mills, L.A. Crompton, A. Bannink, J. Dijkstra, W.J.J. Gerrits, and J. France. 2004. An integrated mathematical model to evaluate nutrient partition in dairy cattle between the animal and its environment. *Animal Feed Science Technology* 112:131–154.

Kebreab, E., J. France, B.W. McBride, N.E. Odongo, A. Bannink, J.A.N Mills, and J. Dijkstra. 2006a. Evaluation of models to predict methane emissions from enteric fermentation in North American dairy cattle. In *Nutrient Utilization in Farm Animals: Modelling Approach,* ed. by E. Kebreab, J. Dijkstra, W.J.J. Gerrits, A. Bannink, and J. France, 299–313. Wallingford, UK: CAB International.

Kebreab, E., K. Clark, C. Wagner-Riddle, and J. France. 2006b. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. *Canadian Journal of Animal Science* 86:135–158.

Kebreab, E., K.A. Johnson, S.L. Archibeque, D. Pape, and T. Wirth. 2008. Model for estimating enteric methane emissions from U.S. cattle. *Journal of Animal Science* 86:2738–2748.

Kelly, J.M., B.mKerrigan, L.P. Milligan and B.W. McBride. 1994. Development of a mobile, open circuit indirect calorimetry system. *Canadian Journal of Animal Science* 74:65–72.

Kendall, D.C., K.M. Lemenager, B.T. Richert, A.L. Sutton, J.W. Frank, B.A. Belstra, and D. Bundy. 1998. Effects of intact protein diets versus reduced crude protein diets supplemented with synthetic amino acids on pig performance and ammonia levels in swine buildings. Presented at Purdue University Swine Day, September 3.

Kephart, K.B., G.L. Greaser, J.K. Harper, and H.L. Moore. 2001. Swine production. Pennsylvania State University. http://agalternatives.aers.psu.edu/Publications/swine.pdf.

Kerr, B.J. 1995. Nutritional strategies for waste reduction management. In *New Horizons in Animal Nutrition and Health,* ed. by J.B. Longnecker and J.W. Spears. 47–68. Chapel Hill, NC: Institute of Nutrition.

Kirchgeßner, M., W. Windisch, and H.L. Müller. 1995. Nutritional factors for the quantification of methane production. In *Ruminant Physiology: Digestion, Metabolism, Growth and Reproduction: Proceedings of the Eighth International Symposium on Ruminant Physiology*, ed. by W. Von Engelhardt, S. Leonhard-Marek, G. Breves, and D. Giesecke, 333–348. Stuttgart: Ferdinand Enke Verlag.

Kohn R., and S.W. Kim. 2011. Mechanistic model of enteric methane emissions from ruminants. *Journal of Animal Science* 89 (E-Suppl. 1)/*Journal of Dairy Science* 94 (E-Suppl. 1):261.

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.

Lague, C. 2003. Management practices to reduce greenhouse gas emission from swine production systems. *Advances in Pork Production* 14:287–300.

Lague, C., É. Gaudet, J. Agnew, T.A. Fonstad. 2005. Greenhouse gas emissions from liquid swine manure storage facilities in Saskatchewan. Translated by the American Society of Biological Engineers 48(6):2289–2296.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* 304(5677):1623–1627.

Larson, M.E., M.S. Honeyman, and J.D. Harmon. 2003. Performance and behavior of early-weaned pigs in hoop structures. *Applied Engineering in Agriculture* 19(5): 591–599.

Leip, A., F. Weiss, T. Wassenaar, I. Perez, T. Fellmann, P. Loudjani, F. Tubiello, D. Grandgirard, S. Monni, and K. Biala. 2010. *Evaluation of the Livestock Sector's Contribution to the EU Greenhouse Gas Emissions (GGELS): Final Report.* Prepared by the European Commission, Joint Research Centre, Institute for Environment and Sustainability, Climate Change Unit.

Leng, R.A. 1991. *Improving Ruminant Production and Reducing Methane Emissions from Ruminants by Strategic Supplementation.* EPA Report 400/1-91/004. Washington, DC: Environmental Protection Agency.

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* 40:1383–1394.

Li., W., W.J. Powers, and G.M. Hill. 2011. Feeding distillers dried grains with solubles and organic trace mineral sources to swine and the resulting effect on gaseous emissions. *Journal of Animal Science* TBC:1–14:2010–3611.

Liang, J.B., F. Terada, and I. Hamaguchi. 1989. Efficacy of using the face mask technique for the estimation of daily heat production of cattle. In Energy metabolism of farm animals, ed. by Y. Van Der Honing and W.H. Close, 348–351. Waginingen, The Netherlands: Pudoc.

Lila, Z.A., N. Mohammed, S. Kanda, T. Kamada, and H. Itabashi. 2003. Effect of sarsaponin on ruminal fermentation with particular reference to methane production in vitro. *Journal of Dairy Science* 86:330–336.

Liu, Z., W. Powers, and H. Liu. 2011. *Meta-Analysis of Greenhouse Gas Emissions from Swine Operations.* Paper 1111369. St. Joseph, Michigan: American Society of Biological Engineers.

Lockyer, D.R., and S.C. Jarvis. 1995. The measurement of methane losses from grazing animals. *Environmental Pollution* 90:383–390.

Loh, Z.D. Chen, M. Bai, T. Naylor, D. Griffith, J. Hill, T. Denmead, S.McGinn, and R. Edis. 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.

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.

Luo J., de Klein C.A.M., Ledgard S.F., Saggar S. 2010. Management options to reduce nitrous oxide from intensively grazed pasture: a review. *Agriculture, Ecosystems & Environment* 136: 282–291.

Machmülller, A., D.A. Ossowski, and M. Kreuzer. 2000. Comparative evaluation of the effects of coconut oil, oilseeds and crystalline fat on methane release, digestion and energy balance in lambs. *Animal Feed Science and Technology* 85:41–60.

Madsen, J., B.S. Bjerg, T. Hvelplund, M.R. Weisbjerg, and P. Lund. 2010. Short Communication: Methane and carbon dioxide ratio in excreted air for quantification of the methane production from ruminants. *Livestock Science* 129: 223–227.

Mangino J., K. Peterson, and H. Jacobs. 2003. Development of an emissions model to estimate methane from enteric fermentation in cattle. http://www.epa.gov/ttn/chief/conference/ei12/ green/mangino.pdf.

Mangino, J., K. Peterson, and H. Jacobs. 2003. *Development of an Emissions Model to Estimate Methane from Enteric Fermentation in Cattle.* Washington, DC: Environmental Protection Agency. http://www.epa.gov/ttn/chief/conference/ei12/green/mangino.pdf.

Martin, C., D.P. Morgavi, and M. Doreau. 2010. Methane mitigation in ruminants: From microbe to the farm scale. *Animal* 4:351–365.

Martin, C., J. Rouel, J.P. Jouany, M. Doreau, and Y. Chillard. 2008. Methane output and diet digestibility in response to feeding dairy cows crude linseed, extruded linseed, or linseed oil. *Journal of Animal Science* 86:2642–2650.

Massé, D.I., F. Croteau, N.K. Patni, and L. Masse. 2003. Methane emissions from dairy cow and swine manure slurries stored at 10°C and 15°C. *Canadian Biosystems Engineering* 45:6.1–6.6.

Mathison, G.W., T.A. McAllister, K.J. Cheng, Y. Dong, J. Galbraith, and O. Dmytruk. 1997. Methane emissions from farm animals: Abstract. Presented at the Workshop on Greenhouse Gas Research in Agriculture, Sainte-Foy, Quebec, March 12–14.

Maycher, N. 2003. Climate change central: Greenhouse gas emissions and opportunities for reduction from The Alberta Swine Industry. Discussion Paper C3-012.

Mbanzamihigo, L., V. Fievez, C. da Costa Gomez, F. Piattoni, L. Carlier, and D. Demeyer. 2002. Methane emissions from the rumen of sheep fed a mixed grass-clover pasture at two fertilization rates in early and late season. *Canadian Journal of Animal Science* 82:69–77.

McAllister, T.A., and C.J. Newbold. 2008. Redirecting rumen fermentation to reduce methanogenesis. *Australian Journal of Experimental Agriculture* 48:7–13.

McCarl, B.A., and U.A. Schneider. 2001. Greenhouse gas mitigation in U.S. agriculture and forestry. *Science* 294(5551):2481–2482

McCaughey, W.P., K. Wittenberg, and D. Corrigan. 1997. Methane production by steers on pasture. *Canadian Journal of Animal Science* 77:519–524.

McCaughey, W.P., K. Wittenberg, and D. Corigan. 1999. Impact of pasture type on methane production by lactating beef cows. *Canadian Journal of Animal Science* 79:221–226.

McCrabb, G.J., M. Kurihara, and R.A. Hunter. 1998. The effect of finishing strategy on lifetime methane production for beef cattle in northern Australia. *Proceedings of the Nutrition Society of Australia* 22:55.

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* 82:3346–3356.

McGinn, S.M., Y.H. Chung, K.A. Beauchemin, A.D. Iwaasa, and C. Grainger. 2009. Use of corn distillers' dried grains to reduce enteric methane loss from beef cattle. *Journal of Animal Science* 89:409–413.

McLean, J.A., and G. Tobin. 1987. *Animal and Human Calorimetry.* New York: Cambridge University Press.

Mills, J.A.N, J. Dijkstra, A. Bannink, S.B. Cammell, E. Kebreab, and J. France. 2001. A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: Model development, evaluation, and application. *Journal of Animal Science* 79:1584–1597.

Mills, J.A.N., E. Kebreab, C.M. Yates, L.A. Crompton, S.B. Cammell, M.S. Dhanoa, R.E. Agnew, and J. France. 2003. Alternative approaches to predicting methane emissions from dairy cows. *Journal of Animal Science* 81:3141–3150.

Misselbrook, T.H., D.R. Chadwick, B.F. Pain, and D.M. 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. *Journal of Agricultural Science* 130:183–191.

Moate, P.J., T. Clarke, L.H. Davies, and R.H. Laby. 1997. Rumen gases and load in grazing dairy cows. *Journal of Agricultural Science* 129:459–469.

Moate, P.J., S.R.O. Williams, C. Grainger, M.C. Hannah, E.N. Ponnampalam, and R.J. Ekhard. 2011. Influence of cold-pressed canola, brewers grains and hominy meal as dietary supplements suitable for reducing enteric methane emissions from lactating dairy cows. *Animal Feed Science and Technology* 166–167:254–264.

Moe, P.W., and H.F. Tyrrell. 1979. Methane production in dairy cows. *Journal of Dairy Science* 62:1583–1586.

Moehn, S., J.K.A. Atakora, and R.O. Ball. 2004. Pig diet manipulation: Greenhouse gas emissions from pigs. Presented at the annual meeting of the Canadian Society of Animal Science, Edmonton, Alberta, July 21–23.

Moller, H.B., S.G. Sommer, and B.K. Ahring. 2004. Biological degradation and greenhouse gas emissions during pre-storage of liquid animal manure. *Journal of Environmental Quality* 33:27–36.

Monteny, G.J., A. Bannink, and D. Chadwick. 2006 Greenhouse gas abatement strategies for animal husbandry. *Agriculture, Ecosystems & Environment* 112: 163–170.

Monteny, G.J., C.M. Groenestein, and M.A. Hilhorst. 2001. Interactions and coupling between emissions of methane and nitrous oxide from animal husbandry. *Nutrient Cycling in Agroecosystems* 60:123–132.

Morgavi, D.P., E. Forano, C. Martin, and C.J. Newbold. 2010. Microbial ecosystem and methanogenesis in ruminants. *Animal* 4:1024–1036.

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* 55:279–296.

Murray, B., and L. Olander. 2008. Addressing impermanence risk and liability in agriculture, land use change, and forest carbon projects. In *Mitigation Beyond the Cap: A Series of Briefs on Expanding Climate Mitigation Opportunities.* Durham, NC: Nicholas Institute for Environmental Policy Solutions, Duke University.

Murray, P.J., A. Moss, D.R. Lockyer, and S.C. Jarvis. 1999. A comparison of systems for measuring methane emissions from sheep. *Journal of Agricultural Science* 133:439–444.

Murray, R.M., A.M. Bryant, and R.A. Leng. 1976. Rates of production of methane in the rumen and large intestine of sheep. *British Journal of Nutrition* 36:1–14.

NASS 2010. http://www.nass.usda.gov/index.asp.

NASS. 2011. Charts and maps: Sheep and lambs. http://www.nass.usda.gov/Charts_and_Maps/Sheep_and_Lambs/index.asp.

Newbold, C.J., S. López, N. Nelson, J.O. Ouda, R.J. Wallace, and A.R. Moss. 2005. Proprionate precursors and other metabolic intermediates as possible alternative electron acceptors to methanogenesis in ruminal fermentation in vitro. *British Journal of Nutrition* 94:27–35.

Ni, J.Q., A.J. Heber, T.T. Lim, P.C. Tao, and A.M. Schmidt. 2008. Methane and carbon dioxide emission from two pig finishing barns. *Journal of Environmental Quality* 37:6.

Nicks, B., M. Laitat, F. Farnir, M. Vandenheede, A. Desiron, C. Verhaeghe, and B. Canart. 2004. Gaseous emissions from deep-litter pens with straw or sawdust for fattening pigs. *Animal Science* 78:99–107.

Nicks, B., M. Laitat, M. Vandenheede, A. Desiron, C. Verhaeghe, and B. Canart. 2003. Emissions of ammonia, nitrous oxide, methane, carbon dioxide, and water vapor in the raising of weaned pigs on straw-based and sawdust-based deep litters. *Animal Research* 52:299–308.

Nkrumah, J.D., E.K. Okine, G.W. Mathison, K. Schmid, C. Li, J.A. Basarab, M.A. Price, Z. Wang, and S.S. Moore. 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* 84:145–153.

Odongo, N.E., R. Bagg, G. Vessie, P. Dick, M.M. Or-Rashid, S.E. Hook, J.T. Gray, E. Kebreab, J. France, B.W. McBride. 2007. Long-term effects of feeding monensin on methane production in lactating dairy cows. *Journal of Dairy Science* 90:1781–1788.

Odongo N.E., O. AlZahal, J.E. Las, A. Kramer, B. Kerrigan, E. Kebreab, J. France, and B.W. McBride. 2008. Data capture: Development of a mobile open-circuit ventilated hood system for measuring real-time gaseous emissions in cattle. In *Mathematical Modelling in Animal Nutrition*, ed. by J. France and E. Kebreab, 225–240. Wallingford, UK: CAB International.

Olander, L.P., A.J. Eagle, J.S. Baker, K. Haugen-Kozyra, B.C. Murray, A. Kravchenko, L.R. Henry, and R.B. Jackson. 2011. *Assessing Greenhouse Gas Mitigation Opportunities and Implementation Strategies for Agricultural Land Management in the United States*. Nicholas Institute for Environmental Policy Solutions Report 11-08.

Olsen, K., R. Weidmeier, and R. Banner. 1996. Reducing methane emissions from beef cow herds in range based management systems. In The Livestock Efficiency Program Annual Conf. Proc. October.

Olsen, K. 1997. Reducing methane emissions from beef cow herds in range-based management systems. Ed. M. Gibbs. In *Ruminant Livestock Efficiency Program Annual Conference Proceedings*, August: 46–52.

Osada, T., H.B. Rom, and P. Dahl. 1998. Continuous measurement of nitrous oxide and methane emissions in pig units by infrared photoacoustic detection. *Transactions of the American Society of Agricultural and Biological Engineers* 41:1109–1114.

Owens, L.B., W.M. Edwards, and R.W. Van Keuren. 1989. Sediment and nutrient losses from an unimproved, all-year grazed watershed. *Journal of Environmental Quality* 18:232–238.

Pain, B.F., T.H. Misselbrook, and Y.J. Rees. 1994. Effects of nitrification inhibitor and acid addition to cattle slurry following the surface application or injection to grassland. *Grass Forage Science* 49:209–215.

Pakrou, N., and P. Dillon. 2000. Key processes of the nitrogen cycle in an irrigated and non-irrigated grazed pasture. *Plant and Soil* 224:231–250.

Park, K.H., A.G. Thompson, M. Marinier, K. Clark, and C. Wagner-Riddle. 2006. Greenhouse gas emissions from stored liquid swine manure in a cold climate. *Atmospheric Environment* 40(4):618–627.

Park, K.H., C. Wagner-Riddle, and R.J. Gordon. 2010. Comparing methane fluxes from stored liquid manure using micrometeorological mass balance and floating chamber methods. *Agricultural and Forest Meteorology* 150(2):175–181.

Pattey, E., M.K. Trzinski, and R.L. Desjardins. 2005. Quantifying the reduction in greenhouse gas emissions as a result of composting dairy and beef cattle manure. *Nutrient Cycling in Agroecosytems* 72:173–187.

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):1963–1969.

Pelletier, F., S. Godbout, A. Marquis, L.O. Savard, and J.P. Larouche. 2004. Greenhouse gas and odor emissions from liquid swine manure storage and manure treatment facilities in Quebec. Presented at the ASAE/CSAE annual international meeting, August, 2004, Ottawa, Canada.

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(6): 380–389.

Pepple, L.M., R.T. Burns, H. Xin, H. Li, and J.F. Patience. 2010. A comparison of gaseous emissions from swine finisher facilities fed traditional vs. a DDGS-based diet. Translated by the American Society of Biological Engineers. Presented at the American Society of Biological Engineers conference, Pittsburgh, September 13–16.

Philippe, F.X, M. Laitat, B. Canart, M. Vandenheede, and B. Nicks. 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.

Pinares-Patiño, C.S., R. Baumont, and C. Martin. 2003. Methane emissions by Charolais cows grazing a monospecific pasture of timothy at four stages of maturity. *Canadian Journal of Animal Science* 83:769–777.

Pinares-Patiño, C.S., K.R. Lassey, R.J. Martin, G. Molano, M. Fernandez, S. MacLean, E. Sandoval, D. Luo, and H. Clark. 2011. Assessment of the sulphur hexafluoride (SF_6) tracer technique using respiration chambers for estimation of methane emissions from sheep. *Animal Feed Science and Technology* 166–167:201–209.

Pinares-Patiño, C.S., M.J. Ulyatt, G.C. Waghorn, C.W. Holmes, T.N. Barry, K.R. Lassey, and D.E. Johnson. 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* 140:215–226.

Portejoie, S., J.Y. Dourmad, J. Martinez, and Y. Lebreton. 2004. Effect of lowering crude protein on nitrogen excretion, manure composition, and ammonia emission from fattening pigs. *Livestock Production Science* 91:45–55.

Powers, W.J., S.B. Zamzow, and B.J. Kerr. 2008. Diet modification as a mitigation tool for swine production. Translated by the American Society of Biological Engineers. Presented at the Livestock Environment VIII conference, Iguassu Falls, Brazil, August 31–September 4.

Renewable Fuels Association. 2012. *Industry Resources: Co-products.* http://www.ethanolrfa.org/pages/industry-resources-coproducts.

Reynolds, C.K., J.A.N. Mills, L.A. Crompton, D.I. Givens, and A. Bannick. 2010. Ruminant nutrition regimes to reduce greenhouse gas emissions in dairy cows. In *Energy and Protein Metabolism and Nutrition,* ed. Ed. G.M. Crovetto, 427–437. EAAP No. 127.

Rural Industries Research and Development Corporation (RIRDC). Australian Government. 2009. A Literature Review of Life Cycle Assessment in Agriculture. Publication No. 09/029, Project No. PRJ-002940. S. Harris and V Narayanaswamy.

Rotz, C.A., F. Montes, and D.S. Chianese. 2010. The carbon footprint of dairy production systems through partial life cycle assessment. *Journal of Dairy Science* 93 :1266–1282

Rotz, C.A., M.S. Corson, D.S. Chianese, F. Montes, S.D. Hafner, R. Jarvis, and C.U. Coiner. 2011. *The Integrated Farm System Model: Reference Manual, version 3.4.* University Park, PA: USDAARS Pasture Systems and Watershed Management Research Unit. http://www.ars.usda.gov/SP2UserFiles/Place/19020000/ifsmreference.pdf.

Rumpler, W.V., D.E. Johnson, and D.B. Bates. 1986. The effect of high dietary cation concentrations on methanogenesis by steers fed with or without ionophores. *Journal of Animal Science* 62:1737–1741.

Safley Jr, L.M., and P.W. Westerman. 1988. Biogas production from anaerobic lagoons. *Biological Wastes* 23:181–193.

Safley Jr., L.M., M.E. Casada, J.W. Woodbury, and K.F. Roos. 1992. *Global Methane Emissions from Livestock and Poultry Manure*. Environmental Protection Agency 400/1-91-048.

Schils, R.L.M., A. Verhagen, H.F.M. Aarts, P.J. Kuikman, and L.B.J. Šebek. 2006. Effect of improved nitrogen management on greenhouse gas emissions from intensive dairy systems in the Netherlands. *Global Change Biology* 12:382–391.

Sharpe, R.R., and L.A. Harper. 1999. Methane emissions from an anaerobic swine lagoon. *Atmospheric Environment* 33:3627–3633.

———. 2001. Methane emissions from swine houses in North Carolina. *Chemosphere* 3:1–6.

Sharpe, R.R., L.A. Harper, and F.M. Byers. 2002. Methane emissions from swine lagoons in southeastern U.S. *Agriculture, Ecosystems & Environment* 90:17–24.

Shindell, D., J.C.I. Kuylenstierna, E. Vignati, R. van Dingenen, M. Amann, Z. Klimont, S.C. Anenberg, N. Muller, G. Janssens-Maenhout, F. Raes, J. Schwartz, G. Faluvegi, L. Pozzoli, K. Kupiainen, L. Höglund-Isaksson, L. Emberson, D. Streets, V. Ramanathan, K. Hicks, N.T. Kim Oanh, G. Milly, M. Williams, V. Demkine, and D. Fowler. 2012. Simultaneously mitigating near-term climate change and improving human health and food security. *Science* 335:183–189.

Shores, R.C., D.B. Harris, E.L. Thompson, C.A. Vogel, D. Natschke, R.A. Hashmonay, K.R. Wagoner, and M. Modrak. 2005. Plane-integrated open-path Fourier transform infrared spectrometry methodology for anaerobic swine lagoon emission measurements. *Applied Engineering in Agriculture* 21(3):487–492.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B.A. McCarl, S. Ogle, F. O'Mara, C.W. Rice, B. Scholes, and O. Sirotenko. 2007a. Agriculture. In *Climate Change 2007: Mitigation,* ed. by B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, and L.A. Meyer. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, U.K. and New York: Cambridge University Press.

Smith, P., D. Martino, Z. Cai, D. Gwary, H.H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, R.J. Scholes, O. Sirotenko, M. Howden,T. McAllister, G. Pan, V. Romanenkov, U. Schneider, S. Towprayoon, M. Wattenbach, and J.U. Smith. 2007b. Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society* B.:363.

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.

Stackhouse, K.R., Y. Pan, Y. Zhao, and F.M. Mitloehner. 2011. Greenhouse gas and alcohol emissions from feedlot steers and calves. *Journal of Environmental Quality* 40:899–906.

Steed, J., and G. Hashimoto. 1994. Methane emissions from typical manure management systems. *Bioresource Technology* 50:123–130.

Steinfeld, H., P. Gerber, T. Wassenaar, V. Castel, M. Rosales, and C. de Haan. 2006. *Livestock's Long Shadow: Environmental Issues and Options.* Rome, Italy: Food and Agriculture Organization.

Su, J.J., B.Y. Liu, and Y.C. Chang. 2003: Emission of greenhouse gas from livestock waste and wastewater treatment in Taiwan. *Agriculture Ecosystems & Environment* 95:253–263.

Sun, H., S.L. Trabue, K. Scoggin, W.A. Jackson, Y. Pan, Y. Ahao, I.L. Malkina, J.A. Koziel, and F.M. Mitloehner. 2008. Alcohol, volatile fatty acid, phenol, and methane emissions from dairy cows and fresh manure. *Journal of Environmental Quality* 37:615–622.

Sutton, A.L., K.B. Kephart, M.W.A. Verstegen, T.T. Canh, and P.J. Hobbs. 1999. Potential for reduction of odorous compounds in swine manure through diet modification. *Journal of Animal Science* 77:430–439.

Takahashi, J. 2011. Some prophylactic options to mitigate methane emission from animal agriculture in Japan. *Asian-Australian Journal of Animal Science* 24(2):285–294.

Tedeschi, L.O., D.G. Fox, and T.P. Tylutki. 2003. Potential environmental benefits of ionophores in ruminant diets. *Journal of Environmental Quality* 32:1591–1602.

Thévenot, A., and J. Vayssiéres. 2011. Towards the use of LCA as an approach to evaluate contribution of agriculture to sustainable development. In *Towards Life Cycle Sustainability Management,* ed. by Matthias Finkbeiner. http://www.lcm2011.org/papers.html.

Thoma, G., D. Nutter, R. Ulrich, C. Maxwell, J. Frank, and C. East. 2011. National life cycle carbon footprint study for production of U.S. Swine. Des Moines, Iowa: National Pork Board.

Thoma G., Popp J., Nutter D., et al. 2012. Greenhouse Gas Emissions from Milk Production and Consumption in the United States: A cradle to grave assessment circa 2008. *International Dairy Journal* (special issue: US Dairy GHG Study), in press.

Thomassen, M.A., R. Dalgaard, R. Heijungs, and I.J.M. de Boer. 2008. Attibutional and consequential LCA for milk production. *International Journal of Life Cycle Assessment* 13:339–349.

Thornley, J.H.M., and J. France. 2007. Mathematical models in agriculture, rev. 2nd ed. Wallingford, UK: CAB International.

Tomkins, N.W., and R.A. Hunter. 2004. Methane mitigation in beef cattle using a patented anti-methanogen. *In Proceedings of 2nd Joint Australia and New Zealand Forum on Non-CO_2 Greenhouse Gas Emission from Agriculture,* ed. R. Eckard. Canberra: CRC for Greenhouse Accounting.

UNFAO (United Nations Food and Agricultural Organization). 2000. *The Coming Livestock Revolution.* Background Paper 6. Commission on Sustainable Development, New York.

USDA (U.S. Department of Agriculture). 2011a. U.S. Beef and Cattle Industry: Background Statistics and Information. http://www.ers.usda.gov/news/BSECoverage.htm.

USDA (U.S. Department of Agriculture). 2011b. U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990–2008. Climate Change Program Office, Office of the Chief Economist, U.S. Department of Agriculture Technical Bulletin No. 1930. 159 pp. June 2011.

USDA (U.S. Department of Agriculture). 2007. *Dairy 2007, Part I: Reference of Dairy Cattle Health and Management Practices in the United States.* N480.1007. Fort Collins, CO: USDA, Animal and Plant Health Inspection Services, Veterinary Services, (APHIS-VS), Centers for Epidemiology and Animal Health (CEAH).

———. 2008a. *Beef 2007–08. Part 1: Reference of Cow-Calf Management Practices in the United States, 2007–08*. N512-1008. Fort Collins, CO: USDA, APHIS-VS, CEAH.

———. 2008b. *Swine 2006, Part III: Reference of Swine Health, Productivity, and General Management in the United States, 2006*. N478.0308. Fort Collins, CO: USDA, APHIS-VS, CEAH.

———. 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.

———. 2011. *US Agriculture and Forestry Greenhouse Gas Inventory: 1990–2008.* Technical Bulletin No. 1930. Washington, DC: USDA.

USDA APHIS NAHMS (Animal and Plant Health Inspection Service, National Animal Health Monitoring System). 1997. *Beef Cow/Calf Health and Productivity Audit*. Fort Collins, CO: USDA, APHIS, NAHMS. http://www.aphis.usda.gov/vs/ceah/cahm.

USDA APHIS NAHMS. 2001. *Part I: Reference of Swine Health and Management in the United States*. Fort Collins, CO: National Animal Health Monitoring System.

USDA APHIS NAHMS. 2008. *Reference of 2008 Dairy Management Practices*. Fort Collins, CO: USDA, APHIS, NAHMS. http://www.aphis.usda.gov/vs/ceah/cahm.

USDA NASS (National Agricultural Statistics Service). 1994. *Hogs and Pigs, Final Estimates 1988–92*. Washington, DC: USDA NASS.

USDA NASS. 2004. *Farm and Ranch Irrigation Survey 2003, vol. 3, Special Studies Part 1*. AC-02-SS-1. Washington, DC: USDA NASS.

U.S. EPA (Environmental Protection Agency). 2009. *Inventory of U.S. Greenhouse Gas Emissions and Sinks:1990–2007*. Washington, DC: U.S. EPA. http://epa.gov/climatechange/emissions/downloads09/GHG2007entire_report-508.pdf.

———. 2010. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2008*. Washington, DC: U.S. EPA. http://www.epa.gov/climatechange/emissions/usinventoryreport.html

———. 2011. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2009*. Washington, DC: U.S. EPA. http://www.epa.gov/climatechange/emissions/usinventoryreport.html.

———. 2012. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2010.* Washington, DC: U.S. EPA. http://www.epa.gov/climatechange/emissions/usinventoryreport.html.

Vaddella, V.K., P.M. Ndegwa, H. Joo, and J.L. Ullman. 2010. Impact of Separating Dairy Cattle Excretions on Ammonia Emissions. *Journal of Environmental Quality* 39:1807–1812

Van Amburgh, M.E., L.E. Chase, T.R. Overton, D.A. Ross, E.B. Recktenwald, R.J. Higgs, and T.P. Tylutki. 2010. Updates to the Cornell Net Carbohydrate and Protein System v6.1 and Implications for Ration Formulation. In *Proceedings of the 2010 Cornell Nutrition Conference for Feed Manufacturers,* 144–159.

Van Nevel, C.J., and D.I. Demeyer. 1995. Lipolysis and biohydrogenation of soybean oil in the rumen in vitro: Inhibition by antimicrobials. *Journal of Dairy Science* 78: 2797–2806.

———. 1996. Influence of antibiotics and a deaminase inhibitor on volatile fatty acids and methane production from detergent washed hay and soluble starch by rumen microbes in vitro. *Animal Feed Science and Technology* 37:21–31.

Van Zijderveld, S.M., J. Dijkstra, W.J.J. Gerrits, J.R. Newbold, and H.B. Perdok. 2010. Dietary nitrate persistently reduces enteric methane production in lactating dairy cows. Presented at the Greenhouse Gases and Animal Agriculture Conference, Banff, Canada, October 3–8, 2010.

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* 28:759–766.

Velthof, G.L., J.A. Nelemans, O. Oenema, and P.J. Kuikman. 2005. Gaseous nitrogen and carbon losses from pig manure derived from different diets. *Journal of Environmental Quality* 34:698–706.

Vergé, X.P.C., J.A. Dyer, R.L. Desjardins, and D. Worth, 2007. Greenhouse gas emissions from the Canadian dairy industry during 2001. *Agricultural Systems* 94(3):683–693.

Vergé X.P.C., J.A. Dyer, R.L. Desjardins, and D.E.Worth. 2008. Greenhouse gas emissions from the Canadian beef industry. *Agricultural Systems* 98(2):126–134.

Vergé X.P.C., D.E. Worth, J.A. Dyer, R.L. Desjardins, and B.G. McConkey. 2011. LCA of animal production. In *Green Technologies in Food Production and Processing*, ed. by Joyce I. Boye and Yves Arcand, 83–114. New York: Springer.

Vermoesen, A., O. van Cleemput, and H.G. Hoffman. 1997. Contribution of urine patches to the emission of N_2O. In *Gaseous Nitrogen Emissions from Grassland*, ed. by S.C. Jarvis and B.F. Pain, 189–194. Wallingford, UK: CAB International.

Walmart. 2010. "Walmart Announces Goal to Eliminate 20 Million Metric Tons of Greenhouse Gas Emissions from Global Supply Chain." http://walmartstores.com/pressroom/news/9668.aspx on 2/13/2011.

Westberg, H., B. Lamb, K.A. Johnson, and M.T. Huyler. 2001. Inventory of methane emissions from U.S. cattle. *Journal of Geophysical Research* 106:12,633–12,642.

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: 2402–2414.

Wolin, E.A., R.S. Wolf, and M.J. Wolin. 1964. Microbial formation of methane. *Journal of Bacteriology* 87:993–998.

Wright, A.D.G., P. Kennedy, C.J. O'Neill, A.F. Troovey, S. Popovski, S.M. Rea, C.L. Pimm, and L. Klein. 2004. Reducing methane emissions in sheep by immunization against rumen methanogens. *Vaccine* 22:3976–3985.

Yamulki, S., S.C. Jarvis, and P. Owen. 1998. Nitrous oxide emissions from excreta in a simulated grazing pattern. *Soil Biology & Biochemistry* 30:491–500.

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: 253–263.

Young, B.A., B. Kerrigan, and R.J. Christopherson. 1975. A versatile respiratory pattern analyzer for studied of energy metabolism of livestock. *Canadian Journal of Animal Science* 55:17–22.

Zahn, J.A., J.L. Hatfield, D.A. Laird, T.T. Hart, Y.S. Do, and A.A. DiSpirito. 2001. Functional classification of swine manure management systems based on effluent and gas emission characteristics. *Journal of Environmental Quality* 30:635–647.

Zeeman, G. 1991. Mesophilic and psychrophilic digestion of liquid manure. Ph.D. diss., Wageningen Agricultural University, Wageningen.

Zeman, C., D. Depken, and M. Rich. 2002. Research on how the composting process impacts greenhouse gas emissions and global warming. *Compost Science & Utilization* 10:72–86.

Zhang, Q., X.J. Zhou, N. Cicek, and M. Tenuta. 2007. Measurement of odour and greenhouse gas emissions in two swine farrowing operations. *Canadian Biosystems Engineering* 49:6.13–6.20.

Zimmerman, P., S. Zimmerman, S. Utsumi, and D. Beede. 2011. Development of a user-friendly online system to quantitatively measure metabolic gas fluxes from ruminants. *Journal of Animal Science* 89 (E-Suppl. 1)/*Journal Dairy Science* 94 (E-Suppl. 1), 759.

Zumft, W.G. 1997. Cell biology and molecular basis of denitrification. *Microbiology and Molecular Biology Reviews* 61:533–616.

the Nicholas Institute

The Nicholas Institute for Environmental Policy Solutions at Duke University is a nonpartisan institute founded in 2005 to help decision makers in government, the private sector, and the nonprofit community address critical environmental challenges. The Institute responds to the demand for high-quality and timely data and acts as an "honest broker" in policy debates by convening and fostering open, ongoing dialogue between stakeholders on all sides of the issues and providing policy-relevant analysis based on academic research. The Institute's leadership and staff leverage the broad expertise of Duke University as well as public and private partners worldwide. Since its inception, the Institute has earned a distinguished reputation for its innovative approach to developing multilateral, nonpartisan, and economically viable solutions to pressing environmental challenges.

for more information please contact:

Nicholas Institute for Environmental Policy Solutions
Duke University
Box 90335
Durham, North Carolina 27708
919.613.8709
919.613.8712 fax
nicholasinstitute@duke.edu
nicholasinstitute.duke.edu