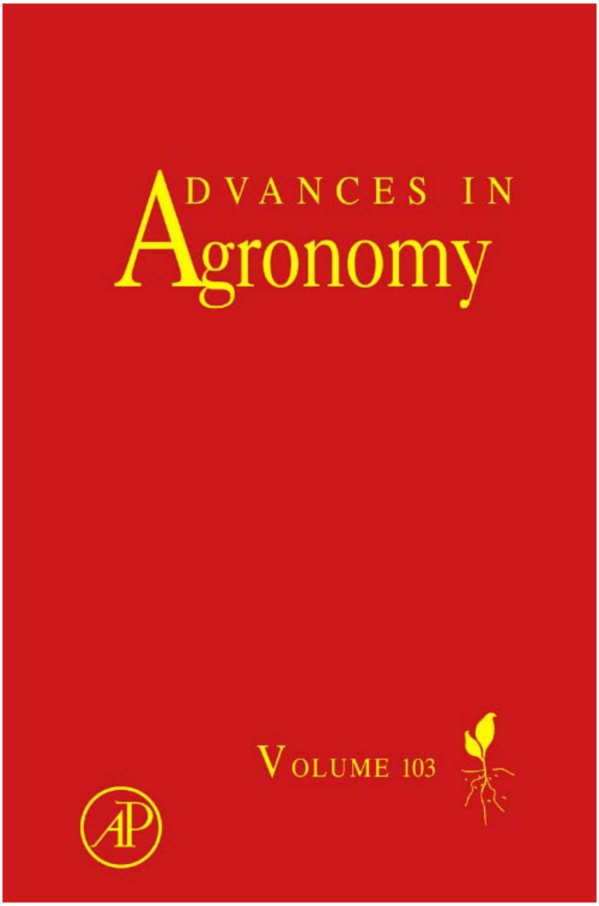

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CHAPTER ONE

Clearing the Air: Livestock's Contribution to Climate Change

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Contents

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Abstract

The United Nations, Food and Agricultural Organization [FAO, Steinfeld, Gerber, Wassenaar, Castel, Rosales, and de Haan (2006). Livestock's Long Shadow. Food and Agriculture Organization of the United Nations] report titled Livestock's Long Shadow (LLS) stated that 18% (approximately 7100 Tg CO_2-eq yr^{-1}) of anthropogenic greenhouse gases (GHGs) are directly and indirectly related to the world's livestock. The report's statement that livestock production is responsible for a greater proportion of anthropogenic emissions than the entire global transportation sector (which emits 4000–5200 Tg CO_2-eq yr^{-1}) is frequently quoted in the public press [Fox News and Kroll (2009). A Tearful, Reluctant Farewell to My Favorite Food: Meat; LA Times (2007). A warming world; pollution on the hoof; livestock emissions are a leading source of greenhouse gases. One solution may be to eat less meat, Los Angeles; NY Times, Op-ed. (2009). Meat and the Planet. New York City] and continues to inform public policy. Recent estimates by the United States Environmental Protection Agency [EPA, Hockstad, Weitz (2009). Inventory of U.S. greenhouse gases and sinks: 1990–2007. Environmental Protection Agency] and the California Energy Commission [CEC—California Energy Commission (2005). Inventory of California Greenhouse Gas Emissions and Sinks: 1990 to 2002 Update] on the impacts of livestock on climate change in the United States and California have arrived at much different GHG estimates associated with direct livestock emissions (enteric fermentation and manure), totaling at less than 3% of total anthropogenic GHG and much smaller indirect emissions compared to the global assessment. Part of the difference of the global versus national predictions is due to the significant weight that has been assigned to the category of "land-use change" patterns related to livestock production (mainly deforestation). Furthermore, LLS attempts a life cycle assessment for global livestock production but does not use an equally holistic approach for its transportation prediction numbers. The primary focus of the present paper is to examine the relative contributions of livestock to climate change at different geographical and production scales. [*Note*:*CO_2equivalents* (CO_2-eq.) represent the total impact (radiative forcing) of GHG in the atmosphere, thereby making it possible to determine the climate change impact of one GHG versus another EPA [EPA and Holtkamp, Irvine, John, Munds-Dry, Newland, Snodgrass, and Williams (2006). "Inventory of U.S. Green House Gases and Sinks: 1996–2006."]. The definition of the *Global Warming Potential* (GWP) for a particular GHG is the ratio of heat trapped by one unit mass of the GHG to that of one unit mass of CO_2 (the GWP of CO_2 is one) over a specific period of time [IPCC (2001). IPCC Third Assessment Climate Change 2001. A Report of the Intergovernmental Panel on Climate Change]. The 100-year GWP for CH_4 and N_2O are 23 times and 296 times the GWP of CO_2, respectively [IPCC (2001). IPCC Third Assessment Climate Change 2001. A Report of the Intergovernmental Panel on Climate Change]. Therefore, for simplicity sake it is common practice to combine the total effects of CO_2, CH_4, and N_2O into CO_2 equivalents (or CO_2-eq).]

1. Introduction

1.1. Overview of global, national, and state (California) reports on livestock's role in climate change

Livestock's Long Shadow (LLS) (FAO *et al.*, 2006) is a life cycle assessment (LCA) of livestock's global impact on biodiversity, land-use, water depletion, water pollution, air pollution, and anthropogenic GHG emissions. The report attempts to quantify the global direct and indirect GHG emissions associated with livestock. Direct and indirect sources of GHG emissions in animal production systems include physiological processes from the animal (enteric fermentation and respiration), animal housing, manure storage, treatment of manure slurries (compost and anaerobic treatment), land application, and chemical fertilizers (Casey *et al.*, 2006; Monteny *et al.*, 2001). Direct emissions refer to emissions directly produced from the animal including enteric fermentation and manure and urine excretion (Jungbluth *et al.*, 2001). Specifically, livestock produce CH_4 directly as a byproduct of digestion via enteric fermentation (i.e., fermenting organic matter via methanogenic microbes producing CH_4 as an end-product) (Jungbluth *et al.*, 2001). Methane and N_2O emissions are produced from enteric fermentation and nitrification/denitrification of manure and urine, respectively (Kaspar and Tiedje, 1981). Previous agricultural estimates have included emissions associated with indirect energy consumption (e.g., electricity requirements, off-site manufacturing, etc.) as five times greater than on-site emissions for cropland production (Wood *et al.*, 2006). Therefore, to accurately estimate the full environmental impact of livestock, indirect emissions need to be quantified. For livestock production, the term indirect emissions refers to emissions not directly derived from livestock but from feed crops used for animal feed, emissions from manure application, CO_2 emissions during production of fertilizer for feed production, and CO_2 emissions from processing and transportation of refrigerated livestock products (IPCC, 1997; Mosier *et al.*, 1998a). Other indirect emissions include net emissions from land linked to livestock including deforestation (i.e., conversion of forest to pasture and cropland for livestock purposes), desertification (i.e., degradation of above ground vegetation from livestock grazing), and release of C from cultivated soils (i.e., loss of soil organic C (SOC) via tilling, natural processes) associated with livestock (IPCC, 1997).

1.2. Global estimates for livestock's impact on climate change

LLS estimates the global contribution of anthropogenic GHG emissions from the livestock sector at 7100 Tg CO_2-eq yr^{-1}, which is approximately 18% of global anthropogenic GHG emissions (FAO *et al.*, 2006). For comparison, global fossil fuel burning accounts for 4000–5200 Tg CO_2-eq yr^{-1} (FAO *et al.*, 2006).

According to FAO *et al.* (2006), the major categories of anthropogenic GHG emissions are:

1. Enteric fermentation and respiration (1800 Tg CO_2-eq yr^{-1})
2. Animal manure (2160 Tg CO_2-eq yr^{-1})
3. Livestock related land-use changes (2400 Tg CO_2-eq yr^{-1})
4. Desertification linked to livestock (100 Tg CO_2-eq yr^{-1})
5. Livestock related release from cultivated soils (230 Tg CO_2-eq yr^{-1})
6. Feed production (240 Tg CO_2-eq yr^{-1})
7. On-farm fossil fuel use (90 Tg CO_2-eq yr^{-1})
8. Postharvest emissions (10–50 Tg CO_2-eq yr^{-1})

Using the first seven of the eight categories listed above, livestock account for 9, 35–40, and 65% of the total global anthropogenic emitted CO_2, CH_4, and N_2O, respectively (FAO *et al.*, 2006).

1.3. United States estimates for livestock's impact on climate change

A second recent report issued by the United States Environmental Protection Agency (EPA) titled "Inventory of United States Greenhouse Gases and Sinks: 1990–2007" (EPA *et al.*, 2007) uses a similar comprehensive LCA methodology compared to LLS (FAO *et al.*, 2006) to characterize the contribution of livestock (and other industries) within the United States with respect to anthropogenic GHG emissions. The EPA *et al.* (2007) report provides a United States national inventory of anthropogenic GHG sources categorized by industry and location (i.e., states within the United States). Based on the total gross anthropogenic emissions of 7150 Tg CO_2-eq yr^{-1} produced within the United States, the EPA calculates that 5.8% (or 413 Tg CO_2-eq yr^{-1}) is associated to the entire agricultural sector (i.e., enteric fermentation, livestock manure management, rice cultivation, agricultural soil management, and burning of crop residues, etc.). Specifically, agriculture in the United States represents 32% of the anthropogenic CH_4 emission and 68% of the N_2O emission (EPA *et al.*, 2009). Within the United States, approximately 198 Tg CO_2-eq yr^{-1} or 2.8% is associated with livestock (i.e., enteric fermentation and manure management).

However, as a reference point for the United States, the transportation sector accounted for 26% (or 1887 Tg CO_2-eq yr^{-1}) of the total (7150 Tg CO_2-eq yr^{-1}) United States anthropogenic GHG portfolio, reflecting the significance of fossil fuel combustion (EPA *et al.*, 2009) and the relative significance of transportation versus animal agriculture. Therefore, the global prediction that livestock account for 18% of GHG emissions and therefore have a "larger" GHG "footprint" than the transportation sector (FAO *et al.*, 2006) is not accurate for the United States.

Within the agricultural sector, the EPA *et al.* (2009) has identified several "key" categories (both direct and indirect sources of GHG emissions). The sources are:

1. Agricultural soil management (209 Tg CO_2-eq yr^{-1})
2. Enteric fermentation (139 Tg CO_2-eq yr^{-1})
3. Manure management (59 Tg CO_2-eq yr^{-1})
4. Rice cultivation (6.2 Tg CO_2-eq yr^{-1})
5. Field burning of agricultural residues (1.4 Tg CO_2-eq yr^{-1})

1.4. California estimates for livestock production effects on climate change

In accordance with EPA and IPCC methods, the state of California compiled its own GHG inventory (CEC, 2005). In 2004, the California inventory estimated that 27 Tg CO_2-eq yr^{-1} or 5.4% of California's gross anthropogenic GHG profile (492 Tg CO_2-eq yr^{-1}) is associated directly and indirectly with agriculture. Within California agriculture, approximately 14 Tg CO_2-eq yr^{-1} or 2.8% is associated with livestock (i.e., enteric fermentation and manure management). Consistent with global (i.e., FAO *et al.*, 2006) and national (i.e., EPA *et al.*, 2009) data, agricultural soil management and enteric fermentation were the greatest emitters of anthropogenic CH_4 and N_2O in California (California Environmental Protection Agency, 2007). As a reference point for California, in 2004 the transportation sector accounted for 182 Tg CO_2-eq yr^{-1} or 37% of the total (492 Tg CO_2-eq yr^{-1}) California anthropogenic GHG portfolio, reflecting the significance of fossil fuel combustion (CEC, 2005) to overall GHG emissions. Again, the global prediction for the relative contribution of livestock versus transportation to climate change (livestock account for 18% of GHG emissions which is more than transportation) is a significantly inaccurate when applied to California, which is the largest dairy and agricultural state within the United States (NASS, 2009).

The major categories of anthropogenic GHG emissions investigated by the State of California (California Environmental Protection Agency, 2007) within the agricultural sector include the following (from highest to lowest emissions):

1. Agricultural soil management (9.1 Tg CO_2-eq yr^{-1})
2. Enteric fermentation (7.2 Tg CO_2-eq yr^{-1})
3. Manure management (6.9 Tg CO_2-eq yr^{-1})
4. Rice cultivation (0.6 Tg CO_2-eq yr^{-1})
5. Field burning of agricultural residues (0.2 Tg CO_2-eq yr^{-1})

While all three reports (CEC, 2005; EPA et al., 2009; FAO et al., 2006) have similar goals (to quantify the relative role of agricultural sources relative to overall anthropogenic GHG emissions), the scope of each report coupled with specific assumptions makes comparison, extrapolation, and interpretation of one report to another cumbersome. These differences are due to several factors including geography (i.e., regional vs global), scope, and methodology (i.e., different assumptions, coefficients, and models). For example, with respect to scope, the EPA et al. (2009) and CEC (2005) reports currently do not identify CO_2 emissions from fossil fuel burning related to agriculture. However, the CEC (2005), EPA et al. (2009), and FAO et al. (2006) reports are largely similar from a methodology perspective.

Figure 1 shows a comparison of predicted relative GHG emissions across all three reports. Globally, FAO et al. (2006) predicts land-use change

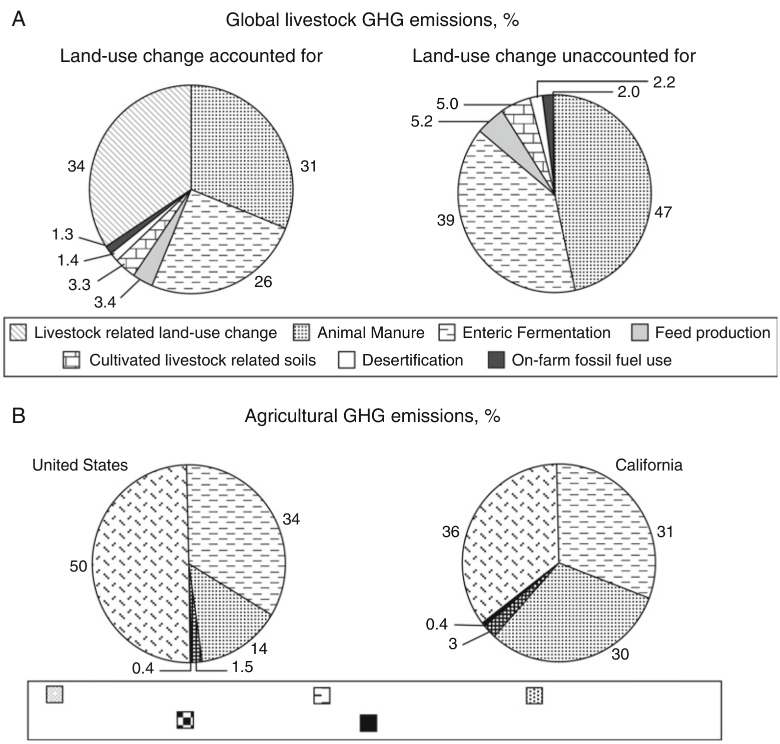

Figure 1 GHG emissions associated with global livestock (A), United States emissions, and California agricultural emissions (B). Direct and indirect N_2O emissions associated with application and deposition of manure are accounted for in the "agriculture soil management" section in the EPA and CEC reports; while in the FAO report, those emissions are accounted for in the animal manure section. Source: data from CEC (2005), EPA et al. (2006), and FAO (2006).

(35.3%) as the primary source of livestock related anthropogenic GHGs (Fig. 1A). The ranking of GHG sources from highest to lowest emissions is identical between EPA *et al.* (2009) and CEC (2005) (Fig. 1B). However, agricultural soil management is a larger source of emissions in the United States as a whole versus California (50.0% vs 36.0%, respectively) (CEC 2005; EPA *et al.*, 2009).

All three reports (CEC, 2005; EPA *et al.*, 2009; FAO *et al.*, 2006) use a combination of Intergovernmental Panel on Climate Change (IPCC) Tier I (uses population data coupled with global emissions factors) and Tier II (same data as Tier I applies more accurate equations based on diet and digestibility coupled with uncertainty analysis). The EPA uses a sophisticated Tier III process-based model (DAYCENT) model to estimate direct emissions from major crops and grassland. The Tier III model uses detailed predictions incorporating local management and weather conditions (among other variables). The Tier I–III models conform the United Nations Framework Convention on Climate Change (IPCC, 2007). However, some differences in assumptions between the three reports were noted:

1. Some parameters were modified to make them more relevant to national and California livestock systems. For example, the State of California adjusted residue-to-crop mass ratio and the fraction of residue applied to reflect the decreased agricultural burning within California (California Environmental Protection Agency, 2007). The EPA report incorporates the Cattle Enteric Fermentation Model (CEFM), which is a refinement of the Tier II calculation (EPA *et al.*, 2009). Major refinements include linkage of livestock performance data to the growth stage of the animal. Specifically, factors such as weight gain, birth rates, pregnancy, feedlot placements, diet, and animal harvest rates are tracked to characterize the United States cattle population on a monthly basis versus the Tier II model, which is updated annually with respect to those variables. Furthermore from a statistical perspective, the EPA report includes a range (e.g., upper and lower boundaries) of emissions estimates predicted by Monte Carlo simulations for a 95% confidence interval (EPA *et al.*, 2009).
2. Another major difference across the three reports is that FAO *et al.* (2006) focuses on livestock while the EPA *et al.* (2009) and California (CEC, 2005) reports include agriculture as a whole (i.e., livestock and plant crops). With respect to the EPA *et al.* (2009) data, it is important to define the agricultural soil management category, which includes applying fertilizers and manure, growing N-fixing crops, retaining crop residues, liming of soils, depositing waste by domestic and grazing animals, and cultivating histosols (i.e., soils with high organic matter content). For example, in the CEC (2004) and EPA *et al.* (2009) reports, agricultural soil management (the largest source of GHG emissions in the United States and California), includes GHG emissions associated with growing fruits, vegetables, fiber grain, as well as livestock pasture and rangeland.

2. Life Cycle Assessment

According to International Standard ISO 14040, an LCA is a "compilation and evaluation of the inputs, outputs, and the potential environmental impacts of a product or service throughout its life cycle" (International Organization for Standardization, 2006). A LCA is a methodology used to assess both the direct and indirect environmental impact of a product from "cradle to grave." Environmental impacts that can be measured include fossil fuel depletion, water use, GWP, ozone depletion, and pollutant production. Figure 2 shows a partial LCA for livestock production (NRC, 2003).

While there are international standards with respect to LCA analysis, uncertainties exist regarding the definitions and "boundaries" of indirect environmental impacts. For example, should the energy required to extract the coal that is used to make the fertilizer, that is applied to the cropland to grow animal feed be included in a "true" LCA of livestock? According to ISO 14040 (International Organization for Standardization, 2006) a comprehensive approach would be ideal but is often not practical. Hence further refinement of the scope and methodology is necessary to increase comparability between LCAs. Lal (2004) described primary (i.e., tilling, sowing, harvesting, pumping water, grain drying), secondary (i.e., manufacturing, packaging, and storing fertilizers and pesticides), and tertiary (i.e., acquisition of raw materials and fabrication of equipment and buildings) emission sources (Lal, 2004). Therefore, based on Lal (2004), one possible method would include LCAs with a numerical suffix indicating the "degree of separation" between the product (e.g., animal protein) and the indirect emissions source input (i.e., the greater the number the more complete and complex the LCA).

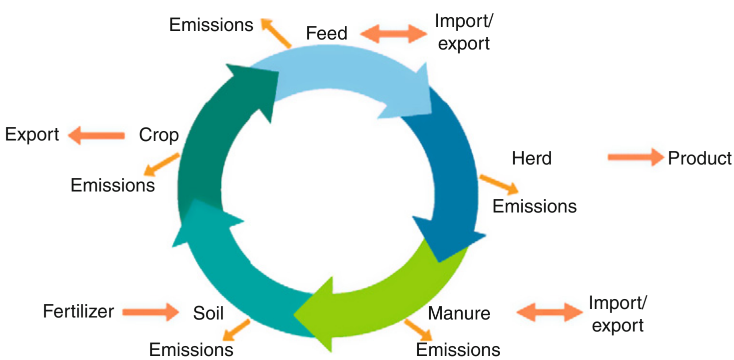

Figure 2 Example of an LCA model for livestock. The model reflects on-site and off-site inputs associated with livestock production. This would not be considered a complete LCA since emissions are only estimated for feed, herd, manure, soil, and crop. Source: NRC (2003).

For example, the LCA in Fig. 2 would be an LCA-1 because only feed, herd, manure, soil, and crop emissions are being accounted for. Regardless, the goal of the LCA is to understand all (or the major) environmental impacts of a product or service to identify the main pollution sources.

Aside from LCA analysis there are several other types of assessment tools for determining the environmental impact of various products and services at a local or global scale. Halberg *et al.* (2005) reviewed multiple assessment tools and concluded that LCAs are ideal for global analysis of products (including livestock production systems (LPSs)) while ecological footprint analysis (EFA) are better suited for studying specific local geographical target areas such as nutrient surplus per hectare (Halberg *et al.*, 2005).

3. Effects of Agriculture on Climate Change

Biogenic emissions of CO_2, CH_4, and N_2O are emitted as part of the natural biogeochemical cycling of C and N (e.g., decomposition or burning of plant material). Anthropogenic emissions of CO_2, CH_4, and N_2O are emitted due to human decisions, activity, and influence of our abiotic and biotic environment (Bruinsma, 2003). Since the industrial revolution in 1750, CO_2 concentrations have increased from 280 to 379 ppm, CH_4 concentrations have increased from 715 to 1732 ppb, and N_2O concentrations have increased from 270 to 319 ppb (IPCC, 1997). Since 1970, atmospheric concentration of CO_2, CH_4, and N_2O has increased by approximately 31, 151, and 17%, respectively, in the United States (USDA, 2004).

Figure 3 shows global CH_4 and N_2O emissions (magnitude and source) within the agricultural sector for 10 different global regions (Smith *et al.*, 2007a). While the gross emissions are not normalized to population (e.g., approximately 20% of the world's population live in developed countries), it is important to recognize that the developing world emits approximately two thirds of all anthropogenic agricultural GHG. In addition, Fig. 3 predicts an increased rate of agricultural emissions through 2020. In six of the 10 world regions, N_2O from soils was the primary agricultural source of GHGs. These N_2O emissions are primarily due to fertilizer and animal manure applied to agricultural soils. In the other four regions (Latin America and the Caribbean, Central and Eastern Europe, the Caucasus and Central Asia, and OECD Pacific), CH_4 from enteric fermentation was the primary source of agricultural emissions (Smith *et al.*, 2007a).

Currently, over half of the total global CH_4 emissions and one third of N_2O emissions are from anthropogenic sources including agriculture, landfills, biomass burning, industrial activities, and natural gas (IPCC, 1997). The IPCC (1997) estimated that the agricultural sector contributes between 10 and 12% of global anthropogenic CO_2 emissions (i.e., fossil fuel

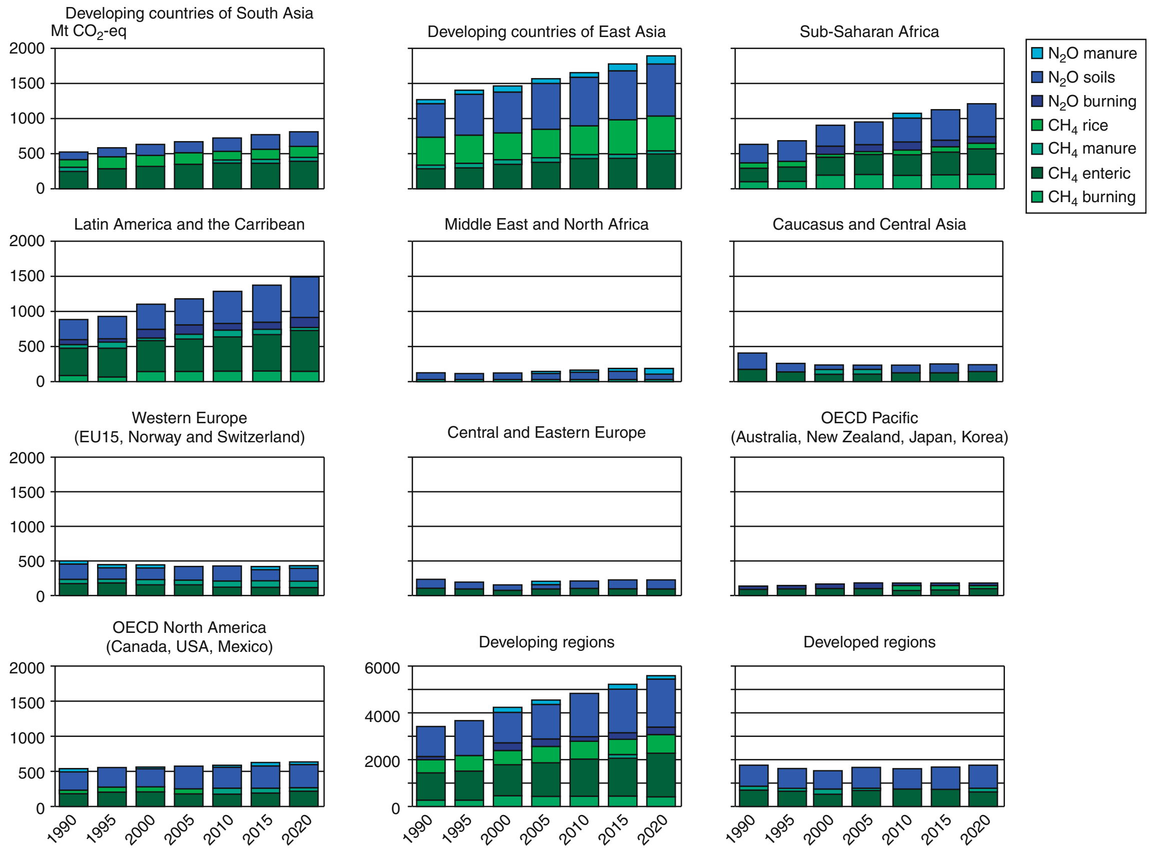

Figure 3 Estimated agricultural N_2O and CH_4 emissions on 10 world regions between 1990 and 2020. Source: Adapted from Fourth Assessment Report of the IPCC (2007) and Smith *et al.* (2007a).

burning), 40% of global anthropogenic CH_4 emissions (i.e., enteric fermentation, wetland rice cultivation, decomposition of animal waste), and 65% of global anthropogenic N_2O emissions (i.e., agricultural soils, use of synthetic and manure fertilizers, manure deposition, biomass burning) (De Gryze *et al.*, 2008; IPCC, 1997). Therefore, agriculture is considered the largest source of anthropogenic CH_4 and N_2O at the global, national, and state level (CEC, 2005; De Gryze *et al.*, 2008; EPA *et al.*, 2009), while transportation is considered the largest anthropogenic source of CO_2 production (EPA *et al.*, 2009).

C and N are part of dynamic cycles that are dependent on multiple environmental conditions. Specifically, oxidation state, pH, water activity, nitrification, denitrification, fermentation, ammonia volatilization, and the microbial ecology of the environment quantitatively and qualitatively affect GHG emissions (CAST, 2004). In addition, emission sources are dispersed and largely driven by biological activity with significant variability over time, space, and management practices (CAST, 2004). Emissions are further affected by local and regional meteorological and soil conditions. Several examples of qualitative variability of GHG production due to environmental conditions have been cited in the literature. For example, under aerobic conditions CO_2 is preferentially produced relative to CH_4 production (De Gryze *et al.*, 2008). However, under anaerobic conditions via methanogenesis (i.e., in rice fields or in a bovine's rumen), CH_4 is preferentially produced relative to CO_2 production. The CH_4 produced can then be converted to CO_2 by microorganisms via CH_4 oxidation (De Gryze *et al.*, 2008). Because CH_4 has 21–23 times the GWP of CO_2, understanding the environmental conditions of CH_4 and CO_2 formation is integral toward both the development of an accurate model and mitigation.

4. Livestock Types and Production Systems

Greenhouse gas emissions from livestock are inherently tied to livestock population size (USDA, 2004). However, due to their greater biomass and unique metabolic function, ruminants are the most significant livestock producer of GHGs (USDA, 2004). Figure 4 shows the estimated global distribution of pigs, poultry, cattle, and small ruminants.

There are currently 1.5 billion cattle and domestic buffalo, and 1.7 billion domestic sheep and goats in the world, which account for over two thirds of the total biomass of livestock (FAO *et al.*, 2006). Within the United States, there are over 94 million beef cattle and 9.3 million dairy cows (NASS, 2009). Cattle are the largest contributing species to enteric fermentation in the United States (EPA *et al.*, 2009). In all three reports discussed in the present chapter (CEC, 2005; EPA *et al.*, 2009; FAO *et al.*, 2006), CH_4 from enteric

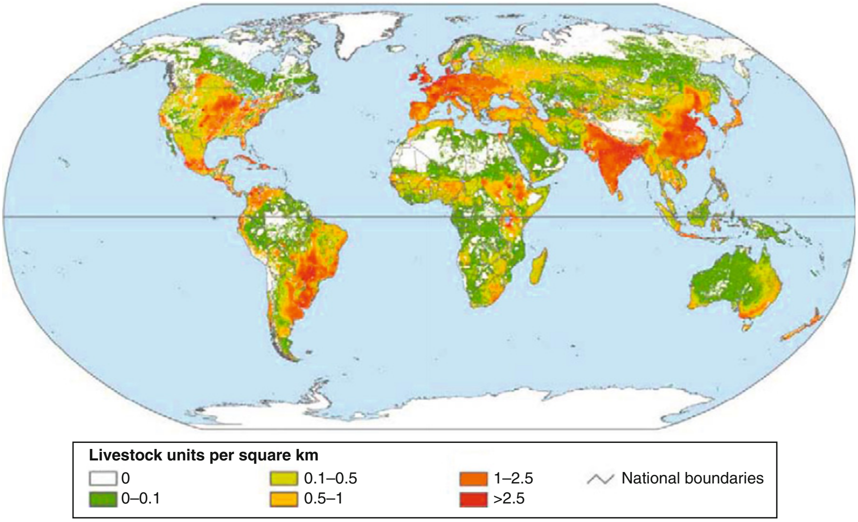

Figure 4 Global estimates of aggregate distribution of pigs, poultry, cattle, and small ruminants (FAO, 2006).

fermentation is the second leading source of GHG from livestock. Therefore, when evaluating LLS (FAO *et al.*, 2006) with respect to GHGs, domesticated ruminants are the primary species studied. However, it is important to recognize the significance of other nonruminant livestock. For example, in the United States swine are the second greatest source of CH_4 and N_2O emissions from manure management and have had a CH_4 and N_2O emissions increase of 34% between 1990 and 2006 (EPA *et al.*, 2006). In addition, pork and poultry production currently consume over 75% of cereal and oil-seed based on concentrate that is grown for livestock (Galloway *et al.*, 2007). Therefore, while ruminants consume 69% of animal feed overall, nonruminates consume 72% of all animal feed that is grown on arable land (Galloway *et al.*, 2007). Consequently, while enteric fermentation from nonruminants is not a significant source of GHG, indirect emissions associated with cropland dedicated to nonruminant livestock might be significant.

The types of LPSs utilized are typically based on socioeconomics, tradition, and available resources. LLS states that extensive (i.e., grazing animals) and intensive (i.e., animals are contained and feed is brought to them) LPSs emit 5000 and 2100 Tg CO_2-eq yr^{-1}, respectively (FAO *et al.*, 2006). While these emissions numbers are not normalized to a per animal unit scale, the type of production system utilized (i.e., landless vs grassland) affects direct (i.e., from the animal) and indirect (i.e., emissions associated with livestock) emissions quantitatively and qualitatively. For example, the low animal density coupled with high land area utilized by extensive systems (e.g., grazing animals occupy 26% of the earth's terrestrial surface) can affect land degradation, deforestation, soil erosion, biodiversity loss, and water contamination (Bruinsma, 2003; FAO *et al.*, 2006). Likewise, because of their high animal density, intensive farming systems can lead to N and P saturation, salinization, and water contamination in addition to reliance on external feed-crop production (Bruinsma, 2003; Mosier *et al.*, 1998a). Therefore, to characterize the GHG ''footprint'' of livestock, the type of LPS needs to be identified and characterized. On the basis of the system parameters (e.g., feed type, animal density, manure storage, and use etc.), FAO *et al.* (2006) divides the LPS into two major types (solely LPSs (L) and mixed farming systems (M)). Figure 5 shows the global distribution of production systems (FAO *et al.*, 2006).

The solely LPSs are further divided into landless LPS (LL) and grassland-based LPS (LG):

1. Landless LPS: Intensive/feedlot type system (defined as systems in which less than 10% of the dry matter fed to animals is farm-produced and where the annual stocking rates are above 10 livestock units per km^2). Developed countries are the primary users of this system with 54.6% of total LL meat production produced in LL systems (FAO *et al.*, 2006). Globally LL-systems account for 75% of the world's broiler poultry supply, 40% of its pork, and over 65% of all poultry eggs (Bruinsma, 2003).

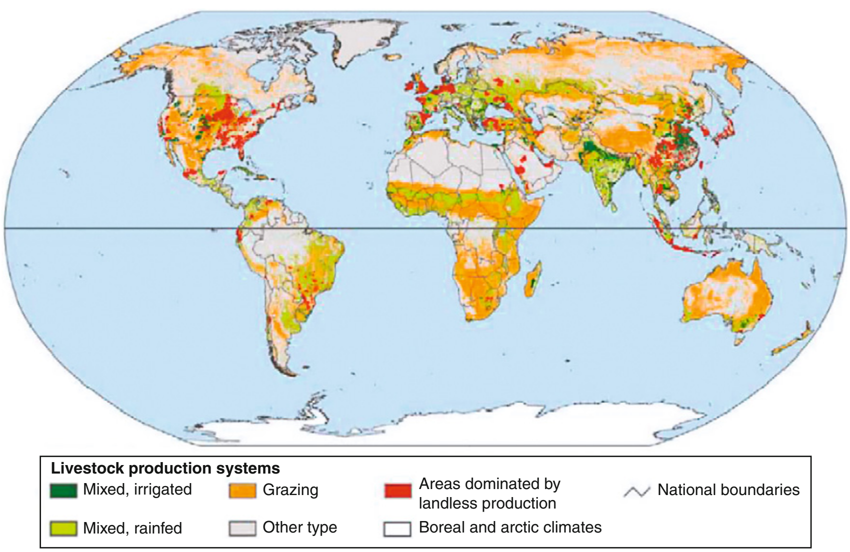

Figure 5 Estimated distribution of livestock production systems. Landless production systems refer exclusively to monogastric production (FAO, 2006).

2. Grassland-based LPSs are defined as areas where more than 10% of dry matter fed to animals is produced at the farm and where annual stocking rates are less than 10 livestock units per hectare of agricultural land (FAO *et al.*, 2006). Grassland-based LPSs are usually present on land that is considered unfit for cropping (primarily semiarid or arid areas). These systems cover the largest global land area and are currently estimated to occupy some 26% of the earth's ice-free land surface (FAO *et al.*, 2006). In South and Central America and part of South East Asia, grazing is often pursued on land cleared from rainforests, where it fuels soil degradation and further deforestation. In semiarid environments, overstocking during dry periods frequently brings risks of desertification (e.g., in sub-Saharan Africa), although it has been shown that marginal pastures do recover quickly if livestock are taken off and rainfall occurs (Bruinsma, 2003). In general, the LG system is characterized by a lower feed quality and a higher feed intake, which leads to higher methane emissions per animal relative to LL production system (Kebreab *et al.*, 2008).

Mixed farming, in which livestock provide manure and power in addition to milk and meat, still predominates for cattle. Mixed farming systems can be divided into the Rain-fed LPS (MR) and the Irrigated LPS (MI).

1. Rain-fed LPS: Mixed systems in which greater than 90% of the value of nonlivestock farm production come from "rain-fed" land use (Ash and Scholes, 2005). In MR, the livestock and cropping components are interwoven. The MR systems are prevalent in temperate, semiarid, and subhumid areas. Approximately two thirds of the total livestock population in India are raised in rain-fed LPS due to the availability of forest grazing and wasteland (Dash and Misra, 2001). These systems typically have large and overstocked livestock populations (Ash and Scholes, 2005). The excess manure is used for cultivation of crops; however, the high animal density can contribute to land-use degradation (Ash and Scholes, 2005).
2. Irrigated mixed farming systems: More than 10% of the value of nonlivestock farm production comes from irrigated land-use. Crop production under irrigated conditions used primarily for rice production with goats as the primary food animal (Ash and Scholes, 2005). Goats typically have low growth and relatively high mortality rates (Ash and Scholes, 2005). Most GHG production is from methane associated with animal manure and irrigated rice cultivation (FAO *et al.*, 2006).

Using the eight categories that most LCA uses to divide anthropogenic GHG emissions associated with global and regional livestock, a comprehensive analysis of each category follows with respect to current literature. Based on the comparison the overall relevancy of each category is then assessed for United States livestock.

5. Enteric Fermentation

Methane production from enteric fermentation is considered the primary source of global anthropogenic CH_4 emissions accounting for approximately 73% of the 80 Tg of CH_4 produced globally per year (Johnson and Johnson, 1995).

Globally as well as in the United States and California, CH_4 released from enteric fermentation accounts for ~1800, 139, and 7 Tg CO_2-eq yr^{-1}, respectively (CEC, 2005; EPA *et al.*, 2009; FAO *et al.*, 2006). LLS (FAO *et al.*, 2006) estimated that 1800 Tg CO_2-eq yr^{-1} is produced globally via CH_4 from enteric fermentation following only land-use change as an emission category.

Ruminants are unique in their ability to convert plants on nonarable land to protein. This characteristic allows ruminants to utilize land and feed that would otherwise be un-used for human food production. At the same time, ruminant livestock is an important contributor to CH_4 in the atmosphere (FAO *et al.*, 2006; IPCC, 2000; USDA, 2004). Methane is produced from the microbial digestive processes of ruminant livestock species such as cattle, sheep, and goats. Nonruminant livestock such as swine, horses, and mules produce less CH_4 than ruminants (USDA, 2004) (Fig. 7).

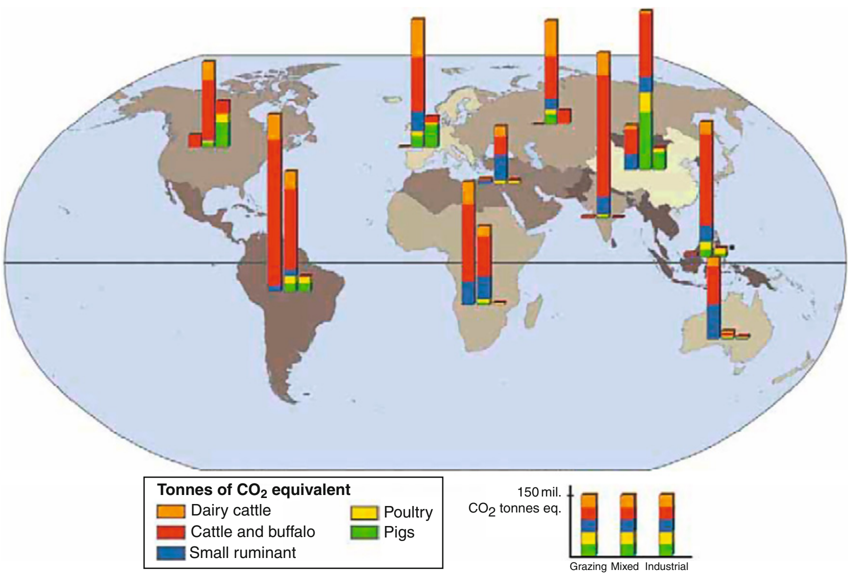

Figure 6 Total GHG emissions from enteric fermentation and manure per species and main productions system (FAO, 2006).

The primary source of CH_4 from ruminant livestock is from the process of enteric fermentation during rumination (Casey *et al.*, 2006; Jungbluth *et al.*, 2001; Kaspar and Tiedje, 1981; Sun *et al.*, 2008). Initial microbial breakdown (essential in ruminant digestion) occurs in the rumen, or large fore-stomach, where microbial fermentation converts fibrous feed into products digested and utilized by the animal (Boadi *et al.*, 2004; USDA, 2004). Rumination promotes digestion of cellulose and hemicellulose through hydrolysis of polysaccharides by microbes and protozoa, which is followed by microbial fermentation generating H_2 and CO_2. Methane is produced as a by-product of enteric fermentation and carbohydrate digestion and is expelled through the mouth via eructation (Monteny *et al.*, 2001).

Global CH_4 emissions are difficult to predict because specific biochemical components of diets are often overlooked in empirical models. Important differences in feed components of the diets used in extensive and intensive LPSs are often overlooked and these systems are viewed as similar. This can result in over- and underestimates of enteric derived CH_4 emissions regionally; especially, where diet components may differ based on the availability of nutrients. Kebreab *et al.* (2008) suggested that IPCC values overestimate CH_4 emissions by 12.5% and underestimate CH_4 emissions by 9.8% for dairy and feedlot cattle, respectively. Mechanistic models might be better suited than empirical models for determining CH_4 emissions as the models are capable of changing source of carbohydrate or addition of fat to decrease methane (Kebreab *et al.*, 2008). Models that predict methane emissions should depend on the diet being fed and the variables relevant to an animal on a particular diet (Ellis *et al.*, 2009). Predictions of CH_4 for an animal on a high grain diet should include some aspect of crude fiber (FC), starch, or forage percentage; while, an animal on a high-fat diet, predictions should include a fat variable (Ellis *et al.*, 2009). In ruminant livestock, enteric fermentation is strongly affected by quantity and quality of their diet (Johnson and Johnson, 1995). Production of CH_4 in ruminants is directly correlated to a loss of metabolizable energy and has been studied in depth during performance studies that aimed at improvements of feed efficiency (Johnson and Johnson, 1995; Jungbluth *et al.*, 2001; Mosier *et al.*, 1998b). Cattle typically lose 2–12% of their ingested energy as eructated CH_4 (Johnson and Johnson, 1995). Many factors affect CH_4 emissions from livestock including feed intake, animal size, diet, growth rate, milk production, and energy consumption (Johnson and Johnson, 1995; Jungbluth *et al.*, 2001). Diet and level of production directly affect CH_4 emission rates (Holter and Young, 1992; Jungbluth *et al.*, 2001; Sun *et al.*, 2008). For example, CH_4 outputs are estimated to range from 3.1 to 8.3% of gross energy intake for dry, non-lactating cows and from 1.7 to 14.9% of gross energy intake for lactating cows (Holter and Young, 1992). Enteric CH_4 ethane emissions per unit of production are highest when feed quality and level of production are low (Crutzen *et al.*, 1986).

Mitigation through improved feed efficiency could reduce CH_4 emissions and result in economic benefits to producers while improving global methane emissions (Johnson and Johnson, 1995). The use of high energy concentrate feed typically used in landless (LL) LPSs results in relatively higher animal production rates (Johnson and Johnson, 1995) and thus less CH_4 emitted per unit of output.

Due to the regional differences in animal species, diets, and production systems, (Figs. 5 and 6) globally it is very difficult to determine accurate CH_4 emissions. Most LL LPSs feed high concentrate diets that meet the specific energy requirements of the animal and thus increase production efficiency, using less resources (feed) to obtain a useable product (meat or milk) in less time. In contrast, extensive (grassland) LPSs, where inputs are less controlled and animals roam freely, feed production efficiency decreases. In other words, these animals require more feed and more time to reach an endpoint that yields useable products.

Emissions from livestock can be mitigated through animal management techniques including nutrition, housing, and waste management (Clemens and Ahlgrimm, 2001; Johnson and Johnson, 1995; Mosier *et al.*, 1998b; Phetteplace *et al.*, 2001; Saggar *et al.*, 2004). Recent work has focused on manipulating the abundance and/or activity of rumen methanogens, to improve the efficiency of ruminant production in an ecologically sustainable way (Wright *et al.*, 2004). One major mitigation technique for CH_4 from livestock is through improvement of production efficiency. For example, in the United States, Capper *et al.* (2009) suggests that continued improvement of management systems and technologies in commercial operations would reduce resource use and environmental impact without sacrificing production. When comparing 1944 with 2007 dairies in the United States, Capper *et al.* (2009) found that modern dairies require 21% of animals, 23% of feedstuffs, 35% of the water, and 10% of the land to produce the same one billion kg of milk. Emissions have also been reduced since 1944; dairies today produce 43% of CH_4 and 56% of N_2O per billion kg of milk (Capper *et al.*, 2009). Management with particular emphasis on improvements of production and reproduction efficiency will likely be among the most viable tools to most significantly reduce environmental impact of livestock systems.

5.1. Carbon dioxide emissions from livestock respiration

The CO_2 from respiration of livestock amounts to ~3000 Tg CO_2-eq yr^{-1} but this CO_2 had previously been absorbed via plants (FAO *et al.*, 2006). According to EPA *et al.* (2006), FAO *et al.* (2006), and the Kyoto Protocol (1997), emissions from livestock are part of continuous cycling biological system where plant matter that had once sequestered CO_2 is consumed by livestock and then released back into the atmosphere by respiration to be

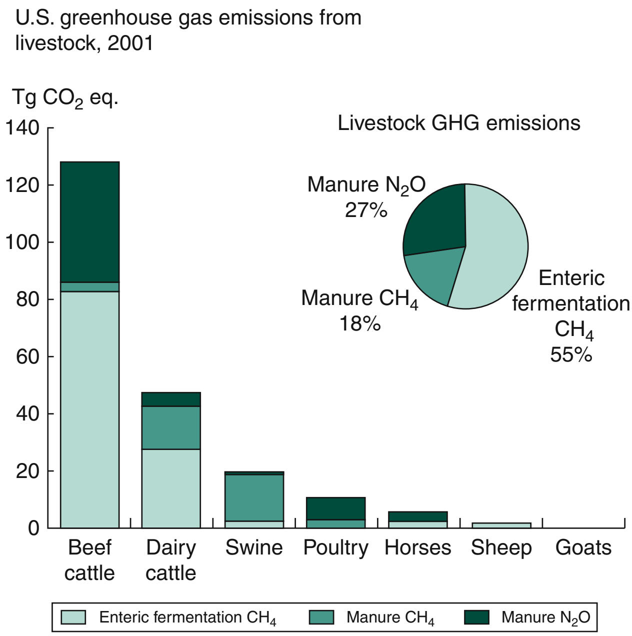

Figure 7 United States greenhouse gas emissions by livestock type, 2001 (USDA, 2004). Note, the United States has approximately 10 times more beef than dairy cattle leading to differences in total contributions (FAO, 2006).

reabsorbed by plants (FAO *et al.*, 2006; Kyoto Protocol, 1997). Consequently, the emitted and absorbed quantities are considered equivalent making livestock a net zero source of CO_2.

6. Animal Manure

The management of animal manure can produce anthropogenic CH_4 via anaerobic decomposition of manure and N_2O via nitrification and denitrification of organic N in animal manure and urine (Bouwman, 1996). LLS (FAO *et al.*, 2006) estimated that global emissions associated with livestock manure (i.e., manure management, manure land application, and indirect manure emissions) total 2160 Tg CO_2-eq yr^{-1}. The EPA *et al.* (2009) and the state of California (CEC, 2005) have assessed that emissions associated with livestock manure (i.e., manure management) in the United States and California total 59.0 and 6.9 Tg CO_2-eq yr^{-1}, respectively. The EPA and CEC place manure land application and indirect manure emissions in the agricultural soil management section. For the EPA and CEC these total 21 and 2.3 Tg respectively.

Typically, when livestock manure is stored or treated in lagoons, ponds, or tanks (i.e., anaerobic conditions), CH_4 emissions are produced in higher amounts than when manure is handled as a solid (e.g., stacks or drylot corrals), or deposited on pasture where aerobic decomposition occurs thereby reducing CH_4 emissions (EPA *et al.*, 2006). Because a strong relationship exists between manure application on land and N_2O emissions (Bouwman, 1996; Jarecki *et al.*, 2008), the emissions associated with fertilization need to be considered a GHG source. However, LLS (FAO *et al.*, 2006) only takes into account emissions from N fertilizer applied to animal feed crops dedicated to food animals, yet including emissions from manure when applied both to animal feed and human crops. The displacement of chemical N fertilizer that is not needed because of N from manure is not considered in LLS. In contrast to chemical fertilizers, the energy input is lower for animal manure (FAO *et al.*, 2006). Therefore, while the direct CO_2-eq kg^{-1} of manure is significantly higher for manure (7–8 kg CO_2-eq kg^{-1} of N) than for fertilizer (between 0.03 and 1.8 CO_2-eq kg^{-1}) (Lal, 2004), the indirect emissions from chemical fertilizer that is not produced need to be accounted for to make an appropriate LCA analysis. Investigating LCAs of GHG emissions associated with fertilizer or manure application on cropland are essential toward understanding the significance of animal manure in agriculture.

A major factor influencing N_2O emissions from agricultural land is N application (Jarecki *et al.*, 2008). The form of fertilizer applied as well as the placement in the soil influences the flux of N_2O emissions (Breitenbeck *et al.*, 1980; Bremner *et al.*, 1981). Both CH_4 and N_2O can be produced by the decomposition of manure. However, N fertilization reduces soil CH_4 oxidation (Jarecki *et al.*, 2008). Methane is produced via the anaerobic decomposition of manure while N_2O is produced via nitrification and denitrification of lad incorporated manure (Chen *et al.*, 2008). Both CH_4 and N_2O production are influenced by multiple variables including climate, soil conditions, substrate availability, and land management practices (Chen *et al.*, 2008). With respect to management in the developed world, the increased use of liquid versus dry manure waste systems (liquid systems produce significantly more methane) in dairy and pig operations has resulted in a relative increase in methane production (FAO *et al.*, 2006). Specifically, in the United States, CH_4 emissions from manure management increased by 34% between 1990 and 2006 primarily due to an increase in liquid manure systems (EPA *et al.*, 2006). One reason for the trend toward liquid-based systems is a response to regulations in the United States including the United States Clean Water Act, which restricts land application rates of manure. The emerging use of CH_4 digesters offers a potential mitigation of CH_4 emissions from liquid manure systems coupled with electricity, gas, and biofuel generation. Current assumptions predict a 50–75% reduction (depending on environmental conditions) in digester GHG emissions from manure when compared with the current system where the manure would otherwise be stored as a liquid slurry in a lagoon (AgStar, 2002).

Wisconsin, New York, Pennsylvania, and California currently have 20, 16, 16, and 15 operating CH_4 digesters, respectively (AgStar, 2002).

Nitrogen assimilation efficiencies vary considerably among different livestock with a range between 10% in beef cattle and 38–75% for swine (Castillo *et al.*, 2001; Hoekstra *et al.*, 2007). As a result, a significant amount of N is returned to the environment through animal excretions (Clemens and Huschka, 2001; Hoekstra *et al.*, 2007). This N can reenter the crop-production cycle, or depending on the conditions be emitted as N_2O or NH_3 (Mosier *et al.*, 1998b). Direct N_2O emissions are produced as part of the N cycle through the nitrification and denitrification of organic N in livestock manure and urine (Mosier *et al.*, 1998b). Annual N losses via N_2O have been previously calculated between 0 and 5% of N applied for manure (Jarecki *et al.*, 2008). Indirect N_2O emissions are produced from N lost as runoff, and leaching of N during treatment, storage, and transportation (Mosier *et al.*, 1998b).

Due to primarily anaerobic conditions of rice production globally, methane production indirectly associated with animal manure application to irrigated rice fields is considered a significant source of emissions. Specifically, due to microbial breakdown of animal manure under anaerobic conditions, global methane emissions account for approximately 60 Tg CO_2-eq yr^{-1} (Verburg and Van der Gon, 2001). In most of the developing world, most rice is grown under these conditions while within the developed world rice is grown with urea as N source.

With respect to animal diet, higher energy feed will have increased methane production from manure. For example, feedlot cattle fed a concentrate diet (i.e., high energy) generate manure with up to 50% higher CH_4 compared to range cattle eating a forage (i.e., low energy) diet (this trend is reversed for enteric fermentation where feedlot versus range cattle produce much less CH_4 per unit of production). Consequently, according to LLS (FAO *et al.*, 2006), the United States (highly intensive production systems) currently has the highest methane emissions factor for manure globally for both dairy and beef cattle (FAO *et al.*, 2006). However, as mentioned earlier (see Section 5.0 on enteric fermentation), high levels of methane emissions from manure management are typically associated with high levels of productivity (FAO *et al.*, 2006). Therefore, per unit of production, more efficient productions systems are superior in the reduction of GHG (Capper *et al.*, 2009).

7. Livestock Related Land-Use Changes

Forests cover approximately 4.1×10^9 ha of the Earth's land area (Dixon *et al.*, 1994) and are estimated to contain 80% of all above ground C and 40% of all below ground terrestrial C (Dixon *et al.*, 1994) (Fig. 8).

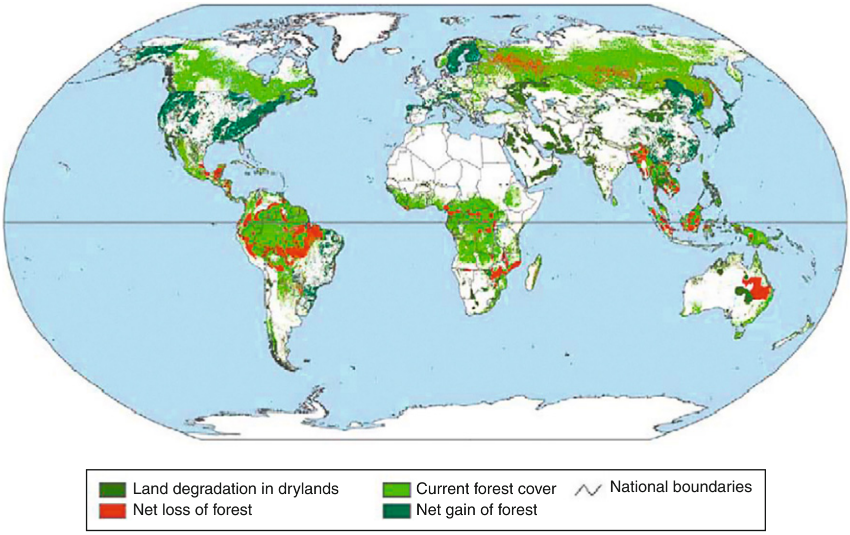

Figure 8 Forest transition and land degradation in dry lands (FAO, 2006).

Russia and Brazil are home to the largest forested areas accounting for 21 and 10% of the total global forestland, respectively (Dixon *et al.*, 1994). High and low latitude forests contain the largest C pools; hence changes (anthropogenic or nonanthropogenic) to specific forested areas can have a greater effect upon on C storage than other forested areas (Dixon *et al.*, 1994).

Land-use change is defined as greenhouse gas emissions from human activities which either change the way land is used (e.g., clearing of forests for agricultural use) or has an effect on the amount of biomass in existing biomass stocks (e.g., forests, village trees, woody savannas, etc.) (IPCC, 2000). From a livestock perspective, land-use changes would include any land adapted for livestock rearing (e.g., animal grazing, production of cropland for livestock feed). Forested areas are particularly sensitive to land-use change. When forest ecosystems undergo relatively abrupt land-use changes, such as deforestation, forest regrowth, biomass burning, wildfires, agriculture abandonment, wetland drainage, plowing, accelerated soil erosion, and so on, a significant loss of SOC and increase in GHG emissions occur (CAST, 2004; Dixon *et al.*, 1994; Houghton *et al.*, 1999).

Using the IPCC's definition of land-use change, livestock uses directly (i.e., pasture, LPS) and indirectly (i.e., production of feed crops) the largest land mass in the world (Bruinsma, 2003; Naylor *et al.*, 2005) and is a primary driver for land-use change. LLS (FAO *et al.*, 2006) estimated that livestock related land-use change produces 2400 Tg CO_2-eq yr^{-1} or 35% of the total GHGs attributed to livestock. LLS (FAO *et al.*, 2006) identifies deforestation in Latin America as the primary source of GHG emissions associated with global livestock. Specifically, land-use changes, including expansion of pasture and arable land for feed crops, primarily occur at the expense of forested land. Forest conversion for permanent crops, cattle ranching, cultivation shifts, and agriculture colonization are considered to contribute equally to the agriculturally driven land-use changes in these countries (Geist and Lambin, 2002). Smith *et al.* (2007b) estimates that over the last 40 years, an average of 6 and 7 Mha of forestland and non-forestland, respectively, was converted to agricultural land in the developing world. Houghton (2003) estimated that "Indonesia and Brazil accounted for approximately 50% of the global land-use change C flux in the 1990s."

While, LLS (FAO *et al.*, 2006) assigned the largest portion of the GHG livestock portfolio to land-use changes, data from EPA *et al.* (2009) show that the United States overall actually increase forestland and that the nation's forests sequester 1078 Tg CO_2-eq yr^{-1} (EPA, 2009). Between 1990 and 2006, the forestland use in the United States increased by 25% from 244 to 304 million hectares (Alig *et al.*, 2003; Smith *et al.*, 2004), resulting in a net uptake in C through trees (EPA, 2009). This gross increase in C sequestration is thought to be related to increased forest area, improved, sustainable timbering (timber growth exceeding harvest), and abandonment of agricultural lands (Alig and Wear, 1992; Alig *et al.*, 1998; Anderson and Magleby, 1997; Flather *et al.*, 1999; Lubowski *et al.*, 2008).

LLS's current LCA methodology (FAO *et al.*, 2006) does not take into account increases in C sinks due to increased management of timberlands in regions like the United States. Forest regeneration, timberland management, and harvesting contribute positively to C sequestration and are highly managed through private landowners. Though harvesting trees as a resource remove much of the aboveground C, there is a positive growth rate of timberlands when it is harvested (Newell and Stavins, 2000). EPA (2009) established through modeling of forest growth that C sequestration is increased if trees are periodically harvested and allowed to regrow rather than maintained as permanently established. For the United States, forest regeneration and expansion is expected to continue and in contrast to some developing countries, deforestation is not a livestock related land-use issue.

In conclusion, LLS (FAO *et al.*, 2006) estimated net C losses associated with converting forested land to grasslands and croplands either directly (pasture) or indirectly through livestock feed production on a global scale. These global predictions result in a significant overestimation of GHG emissions from livestock in developed countries that have established land-use patterns since centuries.

8. Livestock Induced Desertification

Arid and semiarid ecosystems cover greater than 45% of the global land surface (Asner *et al.*, 2003). The most common human agricultural activities on these lands are cattle and sheep grazing/ranching, wood collection, and cultivation (Asner *et al.*, 2003). Desertification (a form of land degradation) primarily occurs in arid, semiarid, and dry subhumid grazing areas (pasture and rangeland) and causes a net loss of C to the atmosphere, ultimately leading to land with reduced biological productivity (Schlesinger *et al.*, 1990). Desertification is generally caused by excessive grazing by livestock, fire, soil erosion, and salinization (Oba *et al.*, 2008). LLS (FAO *et al.*, 2006) estimated that global emissions associated with livestock induced desertification totals 100 Tg CO_2-eq yr^{-1}. These calculations are based upon studies that show a 25–80% decline in SOC in areas with long-term grazing (Asner *et al.*, 2003). Desertification (i.e., land degradation of pasture) is mainly an issue in Africa (2.4 million km^2), Asia (2.0 million km^2), and Latin America (1.1 million km^2) (FAO *et al.*, 2006). The United Nations Environmental Program (UNEP) estimates that 35% of the world's land surface is currently at risk for desertification and more than 20 million hectares are reduced annually to near or complete uselessness (Helldén, 1991).

As mentioned above, nonanimal factors, such as soil erosion and geographical location (higher latitudes may have increased rates of decomposition of soil C), account for some of the SOC losses (Jenkinson, 1991). However, animal factors (i.e., degradation of above ground vegetation) most likely have a more significant contribution to the nonrenewal of decaying organic matter stocks (Asner *et al.*, 2003). Calculating the specific amount that livestock production is responsible for is difficult (FAO *et al.*, 2006). However, livestock do occupy two thirds of the global arable dry land area and the rates of desertification are estimated to be higher in pasture than other land uses (Bruinsma, 2003).

In the United States there are roughly 86 million hectares of federal land grazed by domestic livestock in 17 western states (Bock *et al.*, 1993). Currently, the EPA does not have a desertification category in their inventory of United States GHG emissions and sinks. For the last 150 years, desertification and land degradation in the southwestern United States has led to significant land change (e.g., grassland to shrubland) and to some extend land degradation (Mueller *et al.*, 2007). Historic overgrazing of livestock coupled with climate variation and altered fire regimens are considered some of the drivers of desertification in parts of the South West United States (Mueller *et al.*, 2007; Yanoff and Muldavin, 2008). While grazing of grasslands is considered part of a ruminants natural history, not all grasslands have a symbiotic relationship with grazing ruminates (Bock *et al.*, 1993). These grasslands that are "intolerant" of grazing animals are the most sensitive to desertification and hence SOC loss secondary to overgrazing (Bock *et al.*, 1993). However, according to Loeser *et al.* (2007), areas like the semiarid grasslands of Northern Arizona that have been used at some intermediate level of cattle grazing may be ideal for grazing to maintain native plant diversity. Loeser *et al.* (2007) did not study C emissions or plant biomass (i.e., indicators of C flux); therefore, further research is required to study potential sequestration. However, the concept of livestock being an integral component of ecosystem health is important to recognize.

9. Release from Cultivated Soil

Plowing and tilling coupled with wind, rain, and irrigation exacerbate soil erosion of cropland (Lal, 2004). Approximately 20–30% of SOC is mineralized and released into the atmosphere as CO_2 (Lal, 1999). During the past 40 years, almost one third of the world's cropland has been abandoned due to erosion and degradation (Wood *et al.*, 2006). LLS (FAO *et al.*, 2006) estimated that the loss of C from cultivated soils (i.e., tilling, liming, and emissions related to leguminous feed crops) associated with livestock totals 230 Tg CO_2-eq yr^{-1}. These estimates have a high degree of error based on environment, land management, and annual loss rate coefficients used under those conditions.

As mentioned in the introduction, agricultural soil emissions include non-livestock sources such as emissions associated with production of fruit, vegetables, fiber, grain, as well as livestock and grassland-based emissions. Direct and indirect emissions from agricultural soils related to synthetic N and manure utilization on agricultural soils account for 215 Tg CO_2-eq yr^{-1} in the United States (EPA *et al.*, 2009) and 9.1 Tg CO_2-eq yr^{-1} in California (CEC, 2005). Within the agricultural soil management category, the primary sub-category of emissions for both the U.S. (28%) and California (55%) are emissions associated with synthetic fertilizers. The contribution of the livestock to soil emissions has not been determined in EPA *et al.* (2009). Figure 9 illustrates the sources and pathways of N that result in direct and indirect N_2O emissions.

Greenhouse gas emissions associated with cultivated soils are higher in the United States based on EPA *et al.* (2009) versus the global FAO *et al.* (2006) numbers due to several factors. Before the 2005 "Inventory of United States Greenhouse Gas Emissions" (EPA *et al.*, 2009), GHG estimates within the agricultural sector were based on IPCC emission factors. However, the 2005 inventory includes N_2O emissions using a combination of Tier I and Tier III process-based model (DAYCENT) approaches (Del Grosso *et al.*, 2006). Among other differences, the DAYCENT model includes direct and indirect emissions from agricultural soils due to N additions to cropland and grassland and direct and indirect emissions form

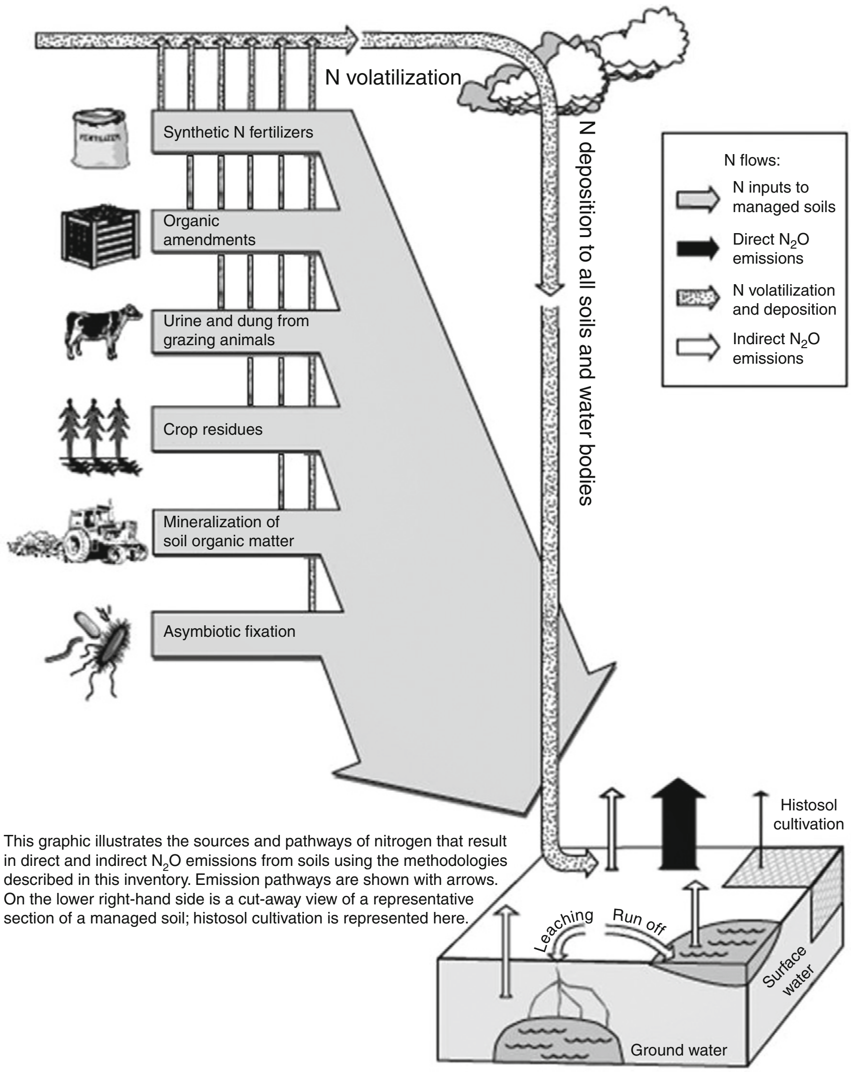

Figure 9 Direct and indirect N_2O emissions from agricultural soils. Sources and pathways of N that result in N_2O emissions from agricultural soil management modified from (EPA *et al.*, 2006).

soils due to the deposition of manure by livestock. In addition, the model is sensitive to inter-annual changes in temperature and management practices. Consequently, the DAYCENT model is considered a more accurate estimate of agricultural N_2O emissions (EPA *et al.*, 2009). In contrast, FAO *et al.* (2006) use IPCC Tier I calculations, which are primarily based on loss of C due to soil erosion.

Cropland versus grassland account for approximately 71 and 29% of total direct anthropogenic GHG emissions from soils, respectively (EPA *et al.*, 2009). Agronomic practices particularly tillage have a significant negative impact on N_2O emissions and SOC losses (CAST, 2004). The N_2O emissions are produced naturally in soils through the microbial processes of nitrification and denitrification (Khalil *et al.*, 2004). Quantitatively the rate of N_2O emissions from soil is highly dependent on several variables including rate of synthetic N-fertilizer application, organic manure application, presence/absence of crop residues, mineralization of soil matter, presence of N-fixing crops, irrigation, and tillage practices (Del Grosso *et al.*, 2006). Consequently, it is important to understand and accurately characterize cropland at high resolution to calculate GHG emissions from cultivated soils. For example, estimates of CO_2 emissions for United States corn, soybean, and wheat production vary from 79 kg C ha^{-1} yr^{-1} for no till soybean to 268 kg C ha^{-1} yr^{-1} for reduced till corn (CAST, 2004).

Agricultural soils and vegetation both emit and sequester C. Therefore, mitigation strategies related to cropping practices are an area of interest. In 2000, the IPCC estimated that conservation tillage can sequester 0.1–1.3 tones C ha^{-1} yr^{-1} globally and could feasibly be adopted on up to 60% of arable lands. Currently the Kyoto protocols do not include C sinks in the emissions inventory for agriculture (Kyoto Protocol, 1997).

10. Carbon Emissions from Feed Production

Historically, most of the resources utilized for livestock nutrition came from the farm itself. While this type of farming is still practiced in some parts of the developing world, most modern livestock operations require a variety of external inputs (i.e., feed production and transport, herbicides, pesticides, etc.) that directly or indirectly utilize fossil fuels and hence produce GHGs (Sainz, 2003). This increased utilization of external inputs allows for increased animal density or intensification of livestock production. In fact, more than half the energy expenditure during livestock production is for feed production (nearly all in the case of intensive beef operations) (FAO *et al.*, 2006). LLS (FAO *et al.*, 2006) estimated that fossil fuel use in manufacturing fertilizer used for animal feed plus emissions associated with application and indirect emissions emits approximately 240 Tg CO_2-eq yr^{-1} globally. Total GHG emissions for mineral fertilizer production are based on synthesis of 14 million tones of mineral fertilizer directly used for fertilization of cropland used solely for animal feed (FAO *et al.*, 2006). The energetic cost of synthetic fertilizer synthesis is between 7 and 65 MJ kg^{-1} of N depending on the fertilizer type and mode of manufacturing (e.g., natural gas versus coal) (FAO *et al.*, 2006). Lal (2004) compiled data estimating C emissions for production, transportation, storage, and transfer of various fertilizers between 0.03 and 1.8 kg CO_2-eq kg^{-1}. The EPA currently does not have a United States domestic value specifically for CO_2 emissions from manufacturing of mineral fertilizer for livestock applications. The only United States numbers currently available are for CO_2 emissions from ammonium manufacture and urea application (13.8 Tg CO_2-eq yr^{-1}) (EPA *et al.*, 2009). Approximately 1% of the world's net energy is utilized in making synthetic fertilizer (Smith, 2002). Carbon dioxide and N_2O emissions are the GHGs associated with the indirect and direct use of fertilizers (Lal, 2004). The primary use of fertilizer in the animal food chain is for the production of corn (FAO *et al.*, 2006). The average corn fertilizer application rate in the United States is 150 kg N ha^{-1} of corn (CAST, 2004). While N_2O emissions occur naturally via nitrification and denitrification, the application of excess N increases the rate of N_2O emissions (Bouwman, 1996). Different rotational farming systems that utilize N-fixing plants before planting corn do not seem to mitigate application rates. This may in part be due to the relatively cheap cost of mineral fertilizer coupled with a "more is better" approach.

Moiser *et al.* (1996) estimated that worldwide application of N as synthetic fertilizer (77.4 Tg yr^{-1}) is in the same range as that of N from manure (77.4 Tg yr^{-1}). Synthetic fertilizers have reduced CH_4 emissions (ammonium nitrate and ammonium sulfate appear to inhibit CH_4 formation) relative to manure, while synthetic fertilizers have relatively higher N_2O emissions. Therefore, attempts to reduce CH_4 from manure sources may well increase other emissions including N_2O (Mosier *et al.*, 1996).

In the United States, mineral fertilizers are the dominant form of crop N supplementation and many semi-developed areas of the world are quickly switching to this model. For example, in the United States in 2001, 10,800 Gg yr^{-1} of N from synthetic fertilizer was used versus 2950 Gg yr^{-1} from livestock manure applied (USDA, 2004). However, it is important to recognize that the synthetic fertilizer produced and utilized in the developed world in general has lower ammonia losses to the environment (4% compared to up to 30% depending on the type of fertilizer and conditions) than the mineral fertilizers used in the developing world (Bouwman, 1996).

Although the use of manure leads to higher direct GHG emissions than mineral fertilizers (Khalil *et al.*, 2008), data comparing net direct and indirect emissions was not incorporated into the LCA of LLS (FAO *et al.*, 2006). In addition, as previously noted in Section 6.0, while LLS addresses the gross GHG emissions produced via production of mineral fertilizer, whereas the potential displacement of synthetic fertilizer production via the "free" production and usage of animal manure is not being discussed. This information will eventually have to be integrated into a more complex (and more accurate) LCA model that would account for the flow of energy from fossil fuels to N fertilizer, from N fertilizer to feed, and from feed to animal protein. Instead, only the emissions associated with N-fertilization of food animal crops (1 Tg CO_2-eq yr^{-1} using a N application rate of ~150 kg ha^{-1} of corn) were assessed in LLS (FAO *et al.*, 2006).

Concentrates are a primary component of livestock feed, fed in the developed world. Concentrates comprise roughly 40% of all animal feed in the developed world versus 12% in the developing world (FAO *et al.*, 2006). Overall, 32% of the world's cereal production (the primary concentrate) is consumed by livestock (Bruinsma, 2003). The main crops utilized for feed production for livestock are corn (52% of concentrates), barley (19%), wheat (19%), and sorghum (5%) (Bruinsma, 2003; FAO *et al.*, 2006).

Within the United States, the state of California is unique from an animal nutrition perspective. The diversity of crops grown in California and their adaptability for both human and animal consumption allows the dairy industry to utilize cropland in a "dual" noncompetitive fashion. Crops, such as rice (rice hulls), almonds (almond hulls), and citrus fruits (citrus pulp) to name a few, have multiple uses for both humans and animals. This dual-utilization decreases the "footprint" of total cropland required for animal feed while integrating these "waste" products for animal feed (plant residues are rarely utilized as soil amendments in the developing world). Feeding crop by-products to livestock reduces decomposition of organic material and releases of GHG to the atmosphere. Instead of these "waste" products being underutilized and hence off-gassing methane as part of landfill or even as municipal solid waste (MSW) (Zhao *et al.*, 2008), dairy cows are able to supplement their diet with these products. In this situation the net benefit of having ruminates needs to be further investigated and included in a California specific model.

LLS (FAO *et al.*, 2006) does not address emissions from production of pesticides, herbicides and other amendments commonly added to cropland. However, in intensive systems the combined-energy use for seed and herbicide/pesticide production and fossil fuel for machinery "generally" exceeds that for fertilizer production (Swanton *et al.*, 1996). Lal (2004) conducted a comprehensive review of energy required for production, transportation, and storage of herbicides, insecticides, and fungicides. Means CO_2-eq kg^{-1} for herbicides, insecticides, and pesticides were 6.3, 5.1, and 3.9, respectively, which were higher than all N-based fertilizers investigated (Lal, 2004). Estimates compiling C emissions for production, transportation, storage, and transfer of herbicides, insecticides, and fungicides had average equivalent C emissions higher than fertilizer (Lal, 2004). These numbers are complicated as some research shows that emission factors from production are superseded by net reduction in emissions on the cropland primarily due to no-till farming (Hisatomi *et al.*, 2007).

From a technology perspective it should also be noted that superior genetics and technology have made food animal nutrition more efficient from both a production and GHG perspective. For example, a study regarding bovine somatotropin (BST) hormone calculated that if all the dairy cows in the United States were using BST, the current milk supply could be reduced by 11% fewer cows, who would be fed 9% less feed, that would be produced on 6% less land (Johnson and Johnson, 1995). These reductions translate to 6% less fossil fuel use and 9% less methane production (Capper *et al.*, 2008). A recent study by Capper *et al.* (2009) using National Research Council (NRC) nutrient recommendations demonstrated that modern dairy practices in the United States in 2007 versus those in 1944 required 79% fewer animals, 78% less feedstuffs, 90% less land, and 65% less water to produce one billion kg of milk. In addition, the same study showed a 74% reduction in manure, 56% reduction in CH_4, and 46% reduction in N_2O per billion kg of milk produced in 2007 versus 1944 cows. In 1944, the United States dairy population totaled 25.6 million cows and produced 53 billion kg of milk annually (average milk yield per cow of 2074 kg yr^{-1}) versus 9.2 million cows producing 84.2 billion kg of milk annually (average milk yield of 9193 kg yr^{-1}) in 2007 (Capper *et al.*, 2009). The authors attribute this dramatic increase in production to genetics, nutrition, and management. The average time needed to produce a broiler in the United States has gone from 72 days in 1960 to 48 days in 1995 with a 1.8–2.2 increase in slaughter weight and a 15% decrease in feed conversion ratios (kg feed per kg meat) (Naylor *et al.*, 2005).

11. On-Farm Fossil Fuel Use: Diesel and Electricity

On-farm fossil fuel use is highly dependent on the intensity and type of livestock production and the environment of the farm. Once on the farm, fossil fuels are utilized for tilling, irrigation, sowing, the movement of feed, for control of the environment (i.e., cooling, heating, and/or ventilation), for animal waste collection and treatment (i.e., land application, solid separation), and for transportation of products (Johnson and Johnson, 1995; Lal, 2004; Sainz, 2003). LLS (FAO *et al.*, 2006) estimated that on-farm fossil fuel use emits 90 Tg CO_2-eq yr^{-1}. Equivalent estimated for the United States do not exist currently for CO_2 emissions from on-farm fossil fuel use. However, in an intensive system, on-farm use of fossil fuel often produces greater GHG emissions than those from chemical N fertilizer (Sainz, 2003).

For the assessment of global on-farm fossil fuel use associated with livestock production, LLS (FAO *et al.*, 2006) utilizes a single study by Ryan and Tiffany (1998). FAO *et al.* (2006) then extrapolates intensive farming globally and adjusts based on latitude (e.g., at lower latitudes less energy would be required for corn drying). Specifically, LLS focuses at on-farm energy use for nine different commodities (corn, soybeans, wheat, dairy, swine, beef, turkeys, sugar beets, and sweet corn/peas). The study identifies diesel or liquefied petroleum gas (LPG) as the primary source of energy for on-farm energy use for eight of the nine commodities. Overall, predictions fossil fuel use associated with livestock production are weak globally and nationally. However, studies in the United States and France have shown a decrease in energy use dedicated to agriculture since 1980 (Bonny, 1993; Cleveland, 1995).

12. Postharvest: CO_2 from Livestock Processing

The postharvest system includes processing, distribution (transport and storage), and preparation. LLS (FAO *et al.*, 2006) estimated United States emissions between 10–50 Tg CO_2-eq yr^{-1} based on research done in Minnesota (Ryan and Tiffany, 1998). The EPA *et al.* (2006) report does not address postharvest emissions.

Figure 10 shows energy use by energy source on United States farms between 1965 and 2001. While total energy use has leveled since 1990, output per unit of energy input has increased significantly (USDA, 2004). In addition, the adoption of no-till land management has the secondary benefit of decreasing fuel use on farms.

While postharvest CO_2 relative to the other categories listed is not a major emitter of GHG, the wide range of data available creates some uncertainty. This uncertainty is primarily related to the myriad of value-added food animal by-products combined with multiple food processing technologies. For example, for a simple product such as processed beef, the energetic cost ranges between 0.84 and 5.02 MJ kg^{-1} live weight (Ward *et al.*, 1977).

In addition, differences in types of energy used for electricity (hydroelectric versus coal) affects on the GHG output. From an energy perspective, depending on the efficiency and the product, agriculture represents between 20 and 50% of the energy consumed within the food supply chain (Wood *et al.*, 2006). For example, the state of California's energy portfolio will change based on implementation of Assembly Bill 32 (AB-32). Specifically, by 2020, by law the state of California can only produce 1990 levels of anthropogenic GHGs (California Environmental Protection Agency, 2007). In order to achieve this cap, one third of California's energy portfolio will be renewable compared to roughly 10% currently (California Environmental Protection Agency, 2007). Therefore, while postharvest emissions are a relatively low proportion of total livestock emissions, regional differences in emissions factors are expected.

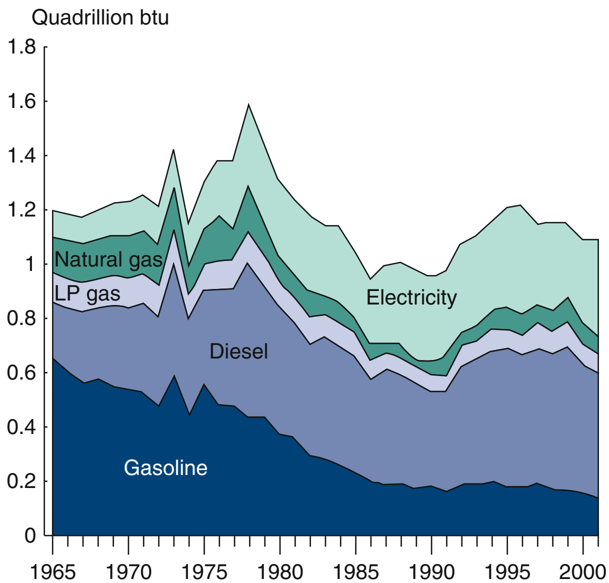

Figure 10 Energy use by agriculture by source 1965–2001 (USDA, 2004).

Postharvest emissions associated with animal feed production and processing of non-food related animal products were not included in FAO *et al.* (2006).

12.1. Transportation

The GHG emissions associated with the transport of animal products ("farm-to-fork") vary according to mode (truck, rail, water) of transport and type of animal product. Previous studies have shown barge to be over eight times more energy efficient than truck and twice as efficient as rail (Rose, 2006). However, these values do not take into account emissions associated with refrigeration for perishable items. LLS (FAO *et al.*, 2006) estimated CO_2 emissions from transport of livestock products to be 0.9 Tg CO_2-eq yr^{-1}. The EPA currently does not measure CO_2 emissions associated with respect to livestock in the United States.

1. When calculating GHG production on a national or regional level, production areas are "assessed" emissions while the receiver region is not assessed any "emissions." These "virtual" emissions are "tallied" solely for the producer and not the consumer. There have been estimates that China's total GHG "footprint" would be reduced by 1/3 if emissions based on usage were calculated instead of emissions based on production (FAO *et al.*, 2006). Likewise, while total CH_4 emissions

from enteric fermentation for Central and South America are approximately one-quarter (486 Tg CO_2-eq yr^{-1}) of the global livestock CH_4, only Central and South America is identified (Steinfeld and Wassenaar, 2007). In a study by Galloway *et al.* (2007), Japan's pig and chicken consumption resulted in the equivalent usage of 50% of Japan's total arable land (Galloway *et al.*, 2007), while trade between Brazil and China is responsible for 15% of the virtual N left behind in Brazil and 20% of Brazil's area to grow soy (Galloway *et al.*, 2007).

2. Reducing "farm-to-fork" transportation emissions does not necessarily reduce GHG emissions from an LCA perspective. If an animal product can be produced internationally in such a way that gross GHG emissions are lower than that same animal product produced under local conditions, then consumption of the product with the shorter "farm-to-fork" distance may in fact have a greater GHG footprint. The point being that the proper integration of international food trade can potentially play an integral role in mitigation of global GHG emissions. While issues related to "food security" encourage local sources of food, the balance between productions for domestic consumption (and food security) needs to be balanced with the "outsourcing" of food production for GHG mitigation.

Within the transportation sector, it is important to discuss the energy and environmental impact of "farm-to-fork" costs for food animals. In the developed world, food animals are often concentrated in landless systems, the transportation of feed grains and other feedstuffs often involves a massive transfer of nutrients between regions. Currently, the EU gets the majority of their soybeans for animal feed from Brazil (Smaling *et al.*, 2008). In the United States, pig operations in the Southeast get the majority of their grain from farms in the mid-west (USDA, 2004). LLS does not assess the transportation or potential environmental costs of these food animal farm-to-fork costs.

12.2. Waste and biomass

To complete a LCA analysis, the waste/use ratio should be determined. Neither FAO *et al.* (2006) nor EPA *et al.* (2006) addresses GHG production due to waste (for animal feed and food animal produced for human consumption). The EPA estimates that 3.6 Tg CO_2-eq yr^{-1} are produced from processing of both meat and poultry from CH_4 emissions associated with industrial waste water (typically anaerobic lagoons) (EPA, 2009). When incorporating these numbers into the United States, the EPA estimates total GHG of the United States agricultural sector to increase from 413 to 417 Tg CO_2-eq yr^{-1}.

The authors were unable to find a specific national or global data on food waste directly related to livestock. However, a study by the USDA Economic Research Service estimated that 2.45 billion kg of edible food at the retail level and 41.36 billion kg at the consumer and foodservice are lost annually accounting for 26% of the total edible food supply (Kantor *et al.*, 1997). This does not include preharvest, on-the-farm, and farm-to-retail losses. Nearly half of the retail losses came from perishable items such as fluid milk and other dairy products and fresh fruits and vegetables (Heller and Keoleian, 2000). No estimates are given on ''wasted'' GHGs produced during production of food that was never consumed. In addition, no data was found on waste streams for food products with a livestock component.

13. Conclusions

With global meat production projected to more than double the current rate by 2050 (Smith *et al.*, 2007b) and the majority of this livestock production growth occurring in the developing world (Wood *et al.*, 2006), assessment of the holistic impacts of food animals in the context of global and regional environmental policy and food security becomes imperative. Much of the growth in the global livestock sector will occur in areas that are currently forested (i.e., parts of South America and South East Asia). It has been well established that significant reductions of carbon sequestering forests will have large effects on global climate change.

LLS (FAO *et al.*, 2006) has been most instrumental in pointing the public attention to the kinds of environmental consequences in which livestock production can potentially result, with special emphasis on climate change. Unfortunately, some of the report's key conclusions (i.e., livestock produces more GHG than transportation) have been applied regionally and out of their intended context, leading to significant consequences on major public policy affairs. For example, the statement that 18% of anthropogenic global GHGs is caused by livestock production and that livestock produces more GHG than transportation (FAO, 2007) is based on inappropriate or inaccurate scaling of predictions, and thus is open to intensive debate throughout the scientific community.

Livestock production in most countries of the developed world (e.g., United States and Europe) has a relatively small GHG contribution within the overall carbon portfolios, dwarfed by large transportation, energy, and other industry sectors. In contrast, livestock production in the developing world can be a dominant contributor to a country's GHG portfolio, due to the developing world's significantly smaller transportation and energy sectors. In the United States, transportation accounts for at least 26% of total anthropogenic GHG emissions compared to roughly 5.8% for all of agriculture, which includes less than 3% associated with livestock production. However, in countries like Paraguay, the trend is likely reversed because of

Paraguay's much smaller transportation and energy sectors, and a relatively large livestock sector, which might contribute to more than 50% of that county's carbon footprint.

The fact that land-use changes associated with livestock (i.e., forested land converted to pasture or cropland used for feed production) are a significant source of anthropogenic GHGs in Latin America and other parts of the developing world is apparent. However, it is likely that any kind of land-use change from the original forestland will lead to great increases in global warming. LLS (FAO *et al.*, 2006) attributes almost half of the climate-change impact associated with livestock to the change of land-use patterns. Latin America has the greatest pool of "unused but suitable" land that is currently covered by forests but could be turned into agricultural crop or livestock production (Bruinsma, 2003). In 2000, Latin America had 203 million hectares arable land in use and 863 million hectares of unused land suitable for cropland (19% in use) (Bruinsma, 2003). Over the same time span, developed countries had 387 million hectares arable land in use and 487 million hectares of unused land suitable for cropland and livestock (44% in use) (Bruinsma, 2003). Transformation of land from forest to agriculture has occurred in the developed countries centuries ago to make way for industrialization and general societal wealth. Not surprisingly, numerous developing countries are currently attempting to develop their economies by turning economically marginal land into production.

The United States and most other developed countries have not experienced significant land-use change practices around livestock production within the last few decades. Instead, over the last 25 years forestland has increased by approximately 25% in the United States and livestock production has been intensified (concentrated geographically), thus reducing its geographical footprint. Modern livestock production has experienced a marked improvement of efficiencies, leading to significantly decreased numbers of animals to produce a given amount product that satisfies the nutritional demands by society (Capper *et al.*, 2009). According to LLS, intensification of livestock production provides large opportunities for climate change mitigation and can reduce greenhouse gas emissions from deforestation, thus becoming a long-term solution to a more sustainable livestock production.

When comparing GHG portfolio sectors such as livestock versus transportation, comparable assessment tools should be used. For example, the transportation figures used in LLS are "direct emissions" associated mainly with combustion during transportation and do not include indirect emissions associated with the transportation or oil industries (i.e., manufacturing of vehicles, resource extraction, etc.). On the other hand, the report assesses livestock holistically from a direct and indirect perspective. A comparison between livestock production versus transportation, with one (livestock) assessment based on a complex LCA and the other (transportation) without LCA, is generally questionable.

Comparing LLS (FAO *et al.*, 2006) with several regional reports (CEC, 2005; EPA *et al.*, 2006) shows large agreement with respect to emission predictions from most livestock related categories. There is general consensus that as a direct GHG category, enteric fermentation in ruminants and manure management are the most important categories within livestock production. Categories like on-farm fuel use or feed production are dwarfed by emissions coming from the animals and their manure.

Many investigators use the international standard (ISO 14040) for LCAs that are often rigid, impractical, and not sufficiently transparent. One means of improvement would be the use of a "numerical suffix system" indicating the "degrees of separation" between the product (e.g., animal protein) and the indirect emissions source input (i.e., the greater suffix number, the more complete the LCA). Furthermore, all current and future assessments of GHG impacts should include mass-balance accounting of energy per GHG unit basis to assess the true environmental impact of direct and indirect emissions. Examples include GHGs associated with displaced fertilizer production through use of animal manure. LLS does not currently account for fertilizer that is not produced because animal manure is present.

LLS (FAO *et al.*, 2006) does not account for "default" emissions. Specifically, if domesticated livestock were reduced or even eliminated, the question of what "substitute" GHGs world be produced in their place has never been estimated. While never explicitly stated in any publication, the idea that if livestock were simply eliminated, 18% of anthropogenic GHGs would also be eliminated as well, is unrealistic. In fact, many of the resources previously dedicated to domesticated livestock would be utilized by other human activities, many of which produce much greater climate change impacts. It is also important to realize that livestock provides not only meat, dairy products and eggs, but also wool, hides, and many other value-added goods and services. Livestock are often closely integrated into mixed and some landless (e.g., landless dairy) farming systems as consumers of crop by-products and sources of organic fertilizer, while larger animals also provide power for plowing and transport. Therefore, to estimate accurately the "footprint" of all livestock, "default" emissions for nonlivestock substitutes need to be estimated and compared to livestock emissions (e.g., manure versus fertilizer, leather versus vinyl, wool versus microfiber, etc.). The net GHG differences between livestock and other land-use forms can then be used to estimate a more accurate GHG "footprint" of livestock's impact.

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