Climate implications of biomass appropriation: Integrating bioenergy and animal feeding systems

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Abstract

Through land use and biomass utilization, humans are dominant forces in the planetary biosphere and carbon and nitrogen cycles. Economic subsidies and policy mandates for producing biomass-sourced fuels and electricity could increase further the human appropriation of planetary net primary productivity. After reviewing the magnitude of organic byproducts available as feedstock, and presenting a model of the climate impact of organic waste management, this dissertation focuses on the climate impact of the main biomass consumers in the United States: livestock, including beef and dairy cattle, chickens (for meat and eggs), pigs and turkeys. Existing estimates of feed consumption by livestock are synthesized, showing that beef cattle in particular are large consumers of cellulosic biomass in the form of hay and grazed roughage. I then determine the extent to which harvesting energy from animal manure can reduce and offset the greenhouse gas (GHG) emissions from producing animal products. Finally, a life cycle assessment (LCA) of an integrated animal product and bioenergy facility is presented. Biomass flows and global warming potential (GWP) are modeled for two systems: one where the animal production and bioenergy facilities are distinct and one where the facilities are integrated. The animal production system includes a mix of animals. Such a system may be able to more efficiently utilize byproducts from each system, but increasing the concentration of animals and manure nutrients may make such a system difficult to implement.

Glossary

AD: Anaerobic digestion

AP: animal products

bioE: Bioenergy (fuels, energy carriers, or energy services supplied with biomass)

CAFO: Concentrated animal feeding operation

Concentrates: Non-roughage **fodder**

DM: Dry matter (non-water part of biomass)

DMx: Dry matter excreted (as manure)

EJ: exajoule (10^{18} joule)

EPA: United States Department of Environmental Protection

ERS: Economic Research Service (of the **USDA**)

EtOH: Ethanol (as fuel)

Fodder: any matter consumed by **livestock**.

GHG: Greenhouse gas

Greenchop ("green"): any **roughage fodder** harvested in a green state, and ensiled, bagged, bunkered, or fed immediately. This term includes traditional "silage" (often corn) and the contemporary, widespread use of bunker and bag **fodder** storage.

GWP: Global warming potential (100 yr timeframe unless specified otherwise)

HANPP: Human appropriation of net primary production

Harvested roughage: any **roughage** not grazed

HHV: Higher heating value

ICDW: industrial, construction & demolition waste

LC: Life cycle

LCA: Life cycle assessment

Livestock: any animal that provides food for, and is managed by, humans.

LUC: Land use change

MJ: megajoule

MSW: municipal solid waste

NASS: National Agricultural Statistics Service (of the **USDA**)

NPP: Net primary production

Roughage: alfalfa and non-alfalfa hays and green-chop, corn and sorghum silage, and grazed biomass **fodder** (or, herbaceous material eaten by a ruminant, including grazed biomass).

US: United States of America

USDA: US Department of Agriculture

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1 Perspectives on biomass

Abstract. A review of biomass use by humans and the greenhouse gas (GHG) implications of using biomass to supply energy services is presented. Human appropriation of global terrestrial net primary productivity is a ratio of appropriation and production; it currently stands at 15-30% of NPP. Over half of used biomass is fed to animals, while equal amounts are eaten directly by humans and burned for energy services. Published estimates of the potential for biomass to supply global energy services vary widely, but a range of 200-500 EJ/ yr has been proposed that considers limitations due to water scarcity, food demand and biodiversity protection. With respect to climate change, uncertainty regarding soil carbon storage capacity, nitrous oxide (N_2O) emissions, and emissions from land use change (LUC) influence the attractiveness of biomass energy (bioE) for GHG abatement. While humans have altered terrestrial biomass patterns for millennia, recent rates of carbon emissions from LUC have been relatively large. In the US, expanded biomass production for bioE could exacerbate the already unsustainable use of water for irrigation, although substantial opportunities exist for increased irrigation efficiency. I calculate that plausible life cycle GHG balances for annual and perennial bioE feedstocks in the Unites States (US) are 14 and 7 g CO_2eq/ MJ, respectively, not including any land use or C stock changes. The influence of biofuel policies on land use change and the degree to which C storage in terrestrial ecosystems can reduce cost of climate change abatement is reviewed. I conclude Chapter 1 with an estimate that appropriating all US biomass exports and all roughage fed to beef cattle would supply about 8 EJ of bioE feedstock.

1.1 *Primary production of biomass*

Autotrophs such as trees and grasses are living organisms which use light (electromagnetic radiation)[1] as an energy source in order to build complex organic materials (e.g. carbohydrates). The organic compounds produced by autotrophs are referred to as "primary production". Gross primary production is a measure of the amount of organic matter produced, and includes the

[1] Some organisms harness energy stored in the chemical bonds of inorganic compounds (e.g. H_2S and NH_3) to function and build new biomass, and use CO_2 as the carbon source. These chemoautotrophes are often found around under-ocean thermal vents and other extreme environments. They are primary producers as long as their inorganic energy sources are produced within the Earth's interior (not biogenic).

compounds consumed by the organism itself for cellular respiration and maintenance. Net primary production (NPP) is a measure of the organic matter from a primary producer that is available to other organisms. The organic matter produced through photosynthesis provides the carbon (C) and energy that is the basis for most life on earth. During photosynthesis, electromagnetic energy is used to liberate electrons and protons from H_2O, releasing O_2 in the process [1]. The liberated electrons and protons eventually allow carbon dioxide (CO_2) to be reduced and organic compounds to be synthesized. Photosynthetic autotrophs (photoautotrophs) produce the vast majority of organic material available for human use, and include terrestrial plants, as well as algae and photosynthetic bacteria [2].

In the most general sense, biomass refers to biological material, either living or dead, that has not gone through geologic transformations (as fossil fuels have). In this work, the term biomass describes any matter that was ultimately sourced from autotrophic organisms not altered by tectonic processes[2]. Some byproduct biomass is animal-based (e.g. waste fat and food scraps), but most of the potential biomass feedstocks (whether byproduct or primary) are plant-based.

The biomass produced by photoautotrophs is crucial for human survival as food eaten directly as plants, or indirectly via animals fed with plant matter. Biomass is also used for clothing, communication (paper), construction, packing and hygienic materials, and in various consumer goods. Biomass has long been used as an energy source by humans. The energy contained in biomass (often released by burning) can be used to keep us warm, to cook, or to power machines, including electricity generators. Biomass may also be processed into more desirable fuels, either

[2] When used in the field of ecology, biomass usually has a more specific definition, referring to the total amount of living biological material in an ecosystem.

solid (e.g. chips, pellets), gaseous (e.g. biogas and syngas) or liquid (e.g. methanol, ethanol, dimethyl ether, or biodiesel).

The efficiency with which plants convert the sun's radiation to biomass is around 2% for crops [3]. The maximum photosynthetic efficiency is reported to be 4.6% for C3 plants, and 6.0% for C4⋆ plants [4]. Typical water requirements (including transpiration) for C3 and C4 plants are about 1000 and 500 moles of water (H_2O) per mole of CO_2 fixed by the plant [3], which correspond to about 700 and 350 kg H_2O/ kg biomass dry matter (DM) for C3 and C4 plants, respectively. These water use efficiencies are consistent with those reported by Berndes [5], who adds that for a crop like miscanthus grown under optimal conditions, 100 kg H_2O/ kg DM represents a likely maximum water use efficiency for a high-yielding crop. Water used for crop irrigation in the United States (US) is reviewed in **Chapter 3**.

1.2 Human appropriation: Biomass as fuel, food, fiber

Humans have large impacts on the Earth's ecosystems [6], and these occur in a wide range of contexts and socio-economic systems [7]. Quantifying the human appropriation of net primary productivity (HANPP) offers a measure of the human influence on the global biosphere. The studies outlined below use somewhat different definitions of HANPP, but these studies typically find between 15% and 30% of global terrestrial NPP is appropriated by humans. In the present work, plant biomass is quantified as moisture-free dry matter (DM), unless otherwise noted, and is assumed to contain 18.5 MJ/ kg DM (higher heating value [HHV]) and 0.47 kg C/ kg DM (adapted from [8-10]).

⋆ C4 plants use a more efficient biochemical mechanism to fix CO_2, thereby reducing photorespiration compared to the more common C3 biochemical pathway.

Annual terrestrial NPP is estimated between 55 and 60 Pg C/ yr globally [9, 11-14]; NPP in the ocean has a similar range, from 48-60 Pg C per yr [11, 12, 15]. C emissions from fossil fuel burning and cement manufacture were 7-8 Pg C [16, 17] in year 2005 (yr2005). The energy content of 60 Pg C of biomass is about 2350 EJ (HHV). For perspective, global human primary energy utilization in yr2005 was 410-490 EJ*, with an average annual increase of 10 EJ for the years 1996-2005 [18]. Thus, energy associated with annual terrestrial NPP is five to six times larger than fossil fuel energy use.

Imhoff et al. [9] report that humans appropriate between 14% and 26% of actual annual terrestrial NPP. This estimate was less than earlier published calculations [19, 20], mainly because Imhoff et al. do not consider below-ground biomass (e.g. roots) as appropriated during livestock grazing. Large uncertainties in appropriation of NPP are associated with the activities that produce (or collect) "vegetal food", wood for fuel, and wood for construction.

Haberl et al. also calculate the fraction of global land-based HANPP, and they estimate both the actual (as of yr2002) and potential (without human influence) NPP [14]. Their calculated total human appropriation is 16 Pg C / yr (24% of the potential NPP [66 Pg C / yr], or 26% of the actual NPP [59 Pg C / yr]). Of this amount, 53% (8.2 Pg C / yr) is appropriated by harvest (including gathering), 40% (6.3 Pg C / yr) by lowered NPP due to human-caused land use change, and 7% (1.1 Pg C / yr) by human-induced fires. For above ground NPP, humans use 29% of potential primary production: 10 of 35 Pg C / yr. Haberl et al. also calculate that the potential NPP of global cropped land is 1.5 times larger than the actual global average

* The primary energy associated with non-thermal electricity generation (e.g. hydro, wind, solar PV) is often calculated using a fossil-plant-based efficiency factor. The range presented represents the result obtained with or without these factors applied.

productivity of croplands. Cropped land, however, usually produces more edible biomass per ha (e.g. grains and fruits) than a “native” ecosystem.

Human manipulation of ecosystems might also increase NPP of a given parcel. Haberl et al. estimate the pre-industrial NPP in the US Midwest was between 12 and 17 Mg DM (including belowground)/ ha/ yr [14]. But current yields of corn in the Midwest are over 25 Mg DM/ ha/ yr, including belowground biomass (assuming 165 bu grain/acre, 15.5% grain moisture, 1:1 ratio of grain biomass to stover DM, and 55:45 ratio of aboveground:belowground biomass [21-24]). While corn yield has increased, the yield potential in the Midwest is difficult to increase [25-28]. In regions where moisture limits plant growth, irrigation can increase yields by an order of magnitude over native NPP. Either way, increased yields are associated with greater evapotranspiration [29]. Irrigation is a large consumptive water use worldwide, but improved water management could perhaps supply increased irrigation demand without increasing the total volume of water used.

Krausmann et al. [30] offer another estimate of global human biomass appropriation: 8.8 Pg C per year (18.7 Pg of DM), or about 16% of actual terrestrial primary production. They do not include human-induced NPP changes, which account for 40% of Haberl’s estimate of HANPP. Economically produced biomass amounted to 5.7 Pg C (12.1 Pg DM) in 2000, of which 12% served as human food, 58% was used as animal feed, 20% as raw material, and 10% as fuel wood. Thus, about the same amount of plant matter was eaten by humans (directly, not including plants eaten by livestock) as was burned by humans as fuel in yr2000 (1.5 Pg DM and 1.2 Pg DM, respectively). The estimated fuel wood use amounts to 22 EJ of energy. In addition, 3.1 Pg

C / yr (6.6 Pg DM) of biomass was appropriated indirectly, including biomass destroyed during harvest (above- and belowground) and human-induced fires. Assuming the energy content of this indirectly appropriated biomass is 15 MJ/ kg DM, the 6.6 Pg DM/ yr amounts to about 100 EJ/ yr. While consumption of traded materials is relatively easy to count because of economic record-keeping, biomass that is collected for home heating and cooking, or through grazing, is more difficult to quantify, as there are no formal records to tally. Thus, estimates for these uses are particularly uncertain.

More than 500 million families worldwide gather biomass to provide residential energy, many of which are in less developed areas [31]. Fernandes et al. estimate that total biofuel consumption globally was stable at 1.2 Pg DM annually between 1850 and 1950, and had increased to 2.46 Pg DM in 2000, 20% of which was used in an industrial setting, while 80% was used residentially [32]. These estimates imply that in yr2000, bioenergy supplied 45.5 EJ, roughly 10% of global primary energy.

1.2.1 Fishing and aquaculture

Captured fish and aquaculture supply a few percent of food energy (calories) globally, but about 15% of all animal-sourced protein is from fishing and aquaculture, and some regional averages are much higher [33]. Humans have either over- or fully-exploited 80% of global fishery resources, while 18% are moderately exploited [34]. Aquaculture of carnivorous species (e.g. salmon) requires inputs of fish-based feed, which puts further strain on ocean systems [35]. Thus, while improved management can prevent a collapse of ocean fisheries [36, 37], the prospect for increasing production from captured ocean fish is somewhat bleak. Increased production of fish worldwide will probably come through aquaculture of herbivorous fish. The feed required for aquaculture could increase demand for terrestrially-sourced biomass.

1.3 Production and supply in the United States

The annual NPP of the terrestrial US (above and belowground) is about 9 Pg DM/yr [38]. Forest and pasture occupy over half the land in the 48 contiguous states (**Figure 1-1**). Land used to grow four major crops (corn, wheat, soybeans and hay) accounts for over 80% of cropped area (**Figure 1-2**). The above-ground plant matter produced in yr2008 by these four crops amounts to approximately one Pg (1 G tonne) of aboveground DM: 0.6 Pg of corn, 0.2 of soybean, and a bit over 0.1 for hay and wheat, each (**Figure 1-3**). In yr2007, about 1.7% of the total mass of forest stock was harvested (about 60% the annual forest growth), corresponding to 0.25 – 0.35 Pg of dry wood biomass [39, 40].

In yr2008, biomass supplied 3.5 EJ in the US, or 3.5% of primary energy consumption in that year. Liquid biofuels and wood-derived industrial boiler-fuel make up two-thirds of the biomass energy (bioE) utilization (**Figure 1-4**) [41]. The aboveground NPP of the four major crops is compared with biomass used for energy in **Figure 1-5**. Total energy supplied via biomass in the US has been relatively consistent over the past two centuries, while population and per capita energy use have increased (**Figure 1-6**).

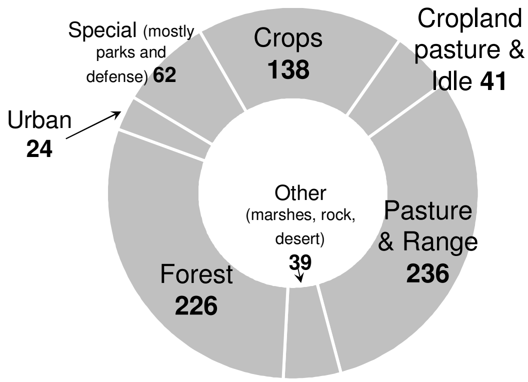

Figure 1-1. Land cover in the 48 contiguous states for the year 2002; units are [M ha]. Total land area is 766 M ha [42].

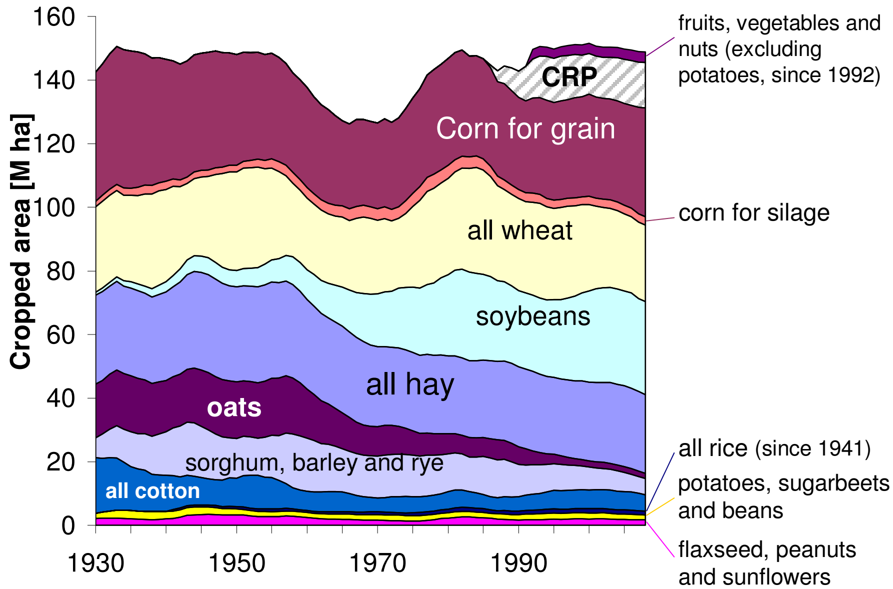

Figure 1-2. Land used for crops in the US, years 1930 through 2008 [43] and land enrolled in the Conservation Reserve Program (CRP) [44]; 5-yr average shown (where applicable). Note that land used for vegetables (excluding potatoes), nuts and fruits is shown only since 1992.

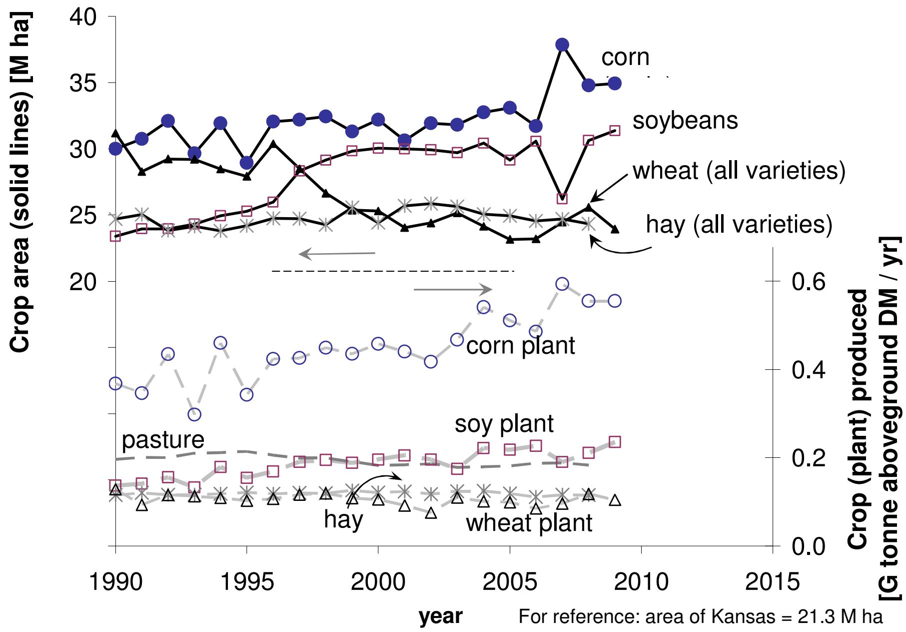

Figure 1-3. Cropped area and aboveground biomass production in US: corn, soybeans, wheat (all varieties), hay (all varieties), and pasture (grazed biomass). Solid lines correspond with the left axis and show area planted with each crop category (except hay, which shows the area harvested). Broken lines show the above ground dry matter (DM) produced by each crop category. Crop harvest and area data are from [43], with grain moisture content from [45]. The ratio of total above ground plant matter to grain is 2 for corn and wheat, and 3 for soybeans. Note the grazed area is not shown since grazing and rangeland area in the US cover about 240 M ha (Figure 1-1). Estimate of grazed biomass from [43], converted from dry corn equivalents using a factor of 1.5 (adapted from [46]).

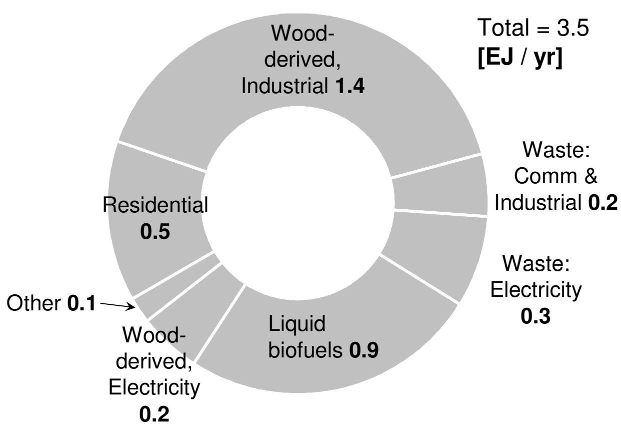

Figure 1-4. Biomass contribution to primary energy supply in US, yr2008, units [EJ/ yr]. Total = 3.5 EJ/ yr [41]. "Liquid biofuels" does not include co-products.

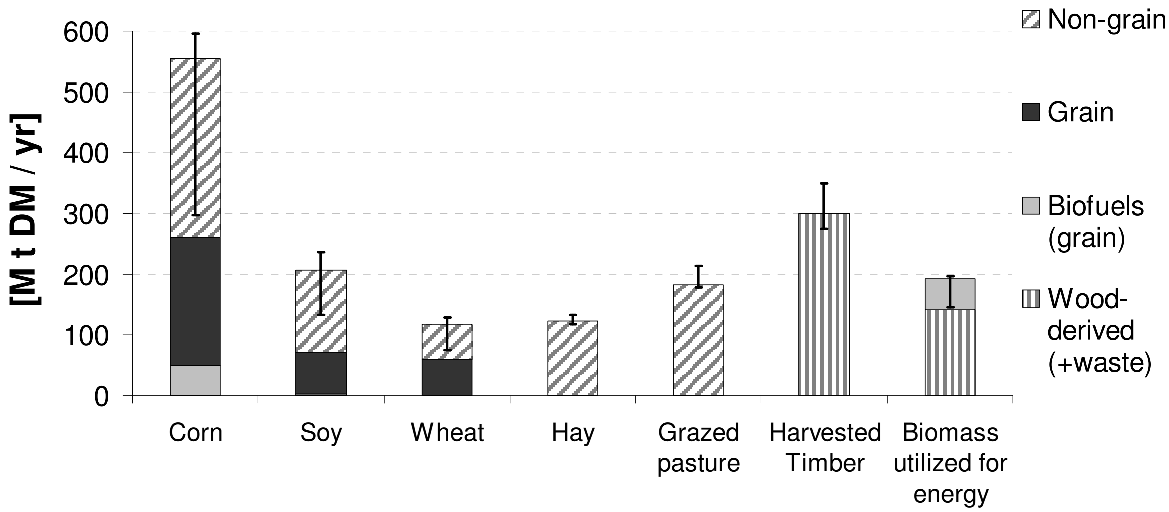

Figure 1-5. Comparison of biomass used annually for energy (left bar) to the annual above ground biomass growth of corn, hay (*includes all varieties and non-corn green-chop), wheat (all varieties), soybean, and grazed pasture. "Biomass utilized for energy" as in Figure 1-4; crops show yr2008 production; ranges show maximum and minimum since the year 1990 (see Figure 1-3). Grain utilized for energy represents ethanol and biodiesel produced from corn grain and soy oil, respectively.

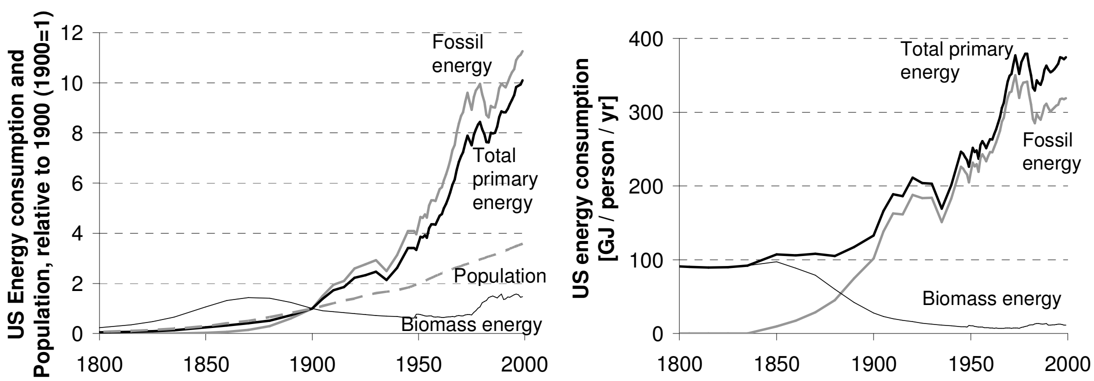

Figure 1-6. Energy use [18] and population [47] in the United States. Left panel shows energy and population, relative to values in yr1900. Right panel shows per capita energy consumption from yr1790 to yr2000. Total primary energy utilization per person has increased since yr1900, both in aggregate and per person. Aggregate biomass energy use has been relatively stable, so per capita biomass energy use has mostly decreased as population has increased.

1.4 *Biomass energy potentials and economics: Review*

> "… there is ample stored solar energy in biomass to meet a very large fraction of our total energy needs."
> --From the mission statement of the Biomass Conversion Research Laboratory, Michigan State University
> [http://www.everythingbiomass.org, accessed June 16, 2009]
>
> "We are not going to run an advanced industrial society on wood chips and cattle dung."
> --Dag Nummedal, Colorado Energy Research Institute
> [As quoted by Barry Friedman in the AAPG EXPLORER, June 2009. American Association of Petroleum Geologists http://www.aapg.org/explorer/2009/06jun/energyfuture0609.cfm, accessed June 8, 2009]

Organic materials include crops, trees, other plants and an array of biomass harvesting and processing byproducts. These are generated domestically in a relatively constant manner. Organic material processing is one of the most important types of activities that humans engage in, with respect to providing bio-physical needs and sanitation. Intended use (i.e. markets) of organic material influences how that matter is handled and processed. For example, grains are often harvested with the degree of storability of the crop as a consideration (weather is, of course, a major factor). Processing may entail post-harvest drying (if moisture is too high for subsequent use or storage), as well as milling, soaking and cooking. Organics also make up a substantial fraction of municipal solid waste (MSW, see **Chapter 2**). Plant-based byproducts in MSW include crop residues, managed landscape trimmings, kitchen and restaurant waste. In the US, about 170 Mt of biomass was utilized for energy services in yr2003, including: 87 in the forest product industry, 32 from urban wood, food and other process residues, 32 for fuel wood, and 5 in bioproducts [48]. In yr2003, the mass of biomass used to produce liquid biofuel was 15 Mt DM per year; in yr2009, it had increased to 60 million t DM per year (not including co-product feed).

Since many of these organic materials have a low economic cost (relative to fuels like petroleum, on an energy basis), they are attractive sources of energy. The historic prices of biomass feedstocks (e.g. corn grain and hay) are lower than petroleum and natural gas on an energy basis. For example, the energy feedstock cost of hay at $100 per tonne is approximately 7 $/GJ, while crude oil at $80 per barrel is about 14 $/GJ. Coal has been an inexpensive fuel - less than 2 $/GJ since yr1985 (yr2005$, from [49]), but a GHG emission price of 50 $/t$CO_2$ would add 4.5 $/GJ to the cost of coal (see **Appendix**, **§1.10**). Thus, biomass as a feedstock for material products and energy services is potentially attractive.

In a review of a possible biofuels future, Ragauskas *et al*. [50] state "Shifting society's dependence away from petroleum to renewable biomass resources is generally viewed as an important contributor to the development of a sustainable industrial society and effective management of greenhouse gas emissions." If there was a general consensus reflected in this statement when it was made in yr2006, it has likely deteriorated with the recognition that land use changes caused by biomass promotion policies could lead to transfers of C from the terrestrial biosphere to the atmosphere, resulting in potentially high emissions rates (see **§1.8**). In addition, possibly high nitrous oxide emissions and uncertainty regarding full soil profile soil carbon content may further dampen prospects for biomass energy to mitigate climate change (see **§1.6**).

Biomass feedstock harvesting from forests and croplands is a geographically dispersed activity. The economic return on using biomass as an electricity or liquid fuel feedstock will vary by region [51], and some regions produce a lot of biomass in the form of crop residues. The decentralized nature of biomass production could possibly be leveraged regionally to provide critical energy services on which society relies and increase resiliency (as described in [52]).

Studies of potential global biomass energy supply report a large range [53]. As discussed above in **§1.2**, the energy contained in global terrestrial NPP (about 2400 EJ/yr) amounts to about six times the global human primary energy utilization (400 to 500 EJ in yr2005). The IPCC offers a range of biomass potential, with values from 40 to greater than 400 EJ/ year globally [54]. These scenarios include projections of yield changes (typically increases) and livestock management practices; the scenarios with higher bioE potentials include conversion of some current agricultural land to bioenergy crops [55, 56]. The IPCC estimates are generally consistent with a recent review of biomass potentials [53]. **Table 1-1** shows a few estimates of available biomass globally. A supply of 2000 Mt of biomass contains roughly 30 EJ of energy (HHV basis, assuming 15 MJ/ kg to account for biomass drying and processing).

Table 1-1. Estimates of global potential of biomass supply [Mt / yr].

	Campbell et al. [57]	Field et al. [8]	Tilman et al. [58]	Hoogwijk et al. [59]	Kim and Dale [60]
Crops on abandoned & degraded land	1700 - 2000	1400 ± 50%	2400	0 - 52000	
Ag residue and waste				1100 – 3200	1600
Forest product residue				500 - 800	
Organic waste				50 - 200	

As with studies of global biomass potential, those which focus on the US also report a wide range of estimates [61]. The US Department of Agriculture (USDA) and the US Department of Energy (DOE) produced a report in yr2005 that outlined the possibility of a one billion tonne (1000 Mt or 1 Pg) supply of biomass for energy in the US [48]. The energy content of 1000 Mt of biomass is about 17 EJ. In yr2003, about 170 Mt of dry biomass were utilized either for energy or as bioproducts. Thus, the magnitude of the biomass energy resource outlined by the

USDA could be over five times the amount being utilized historically. The estimate of total wood resource potential is 370 Mt/yr in the USDA/DOE report, compared to the 130 Mt that were sourced from wood in yr2003. Much of the potential comes from harvesting logging residue, thinning forests for fire management, processing byproducts, and capturing construction and demolition biomass debris. The report estimates that 195 Mt of biomass from cropland are currently available, including 75 Mt of corn stover (the non-grain parts of corn plants) and 35 Mt of manures. Under a hypothetical future scenario with high crop yields, 380 Mt of perennial grass biomass could be available, 445 Mt of crop residues (255 Mt of which are corn stover), and 90 Mt of grain for biofuel production. Under this high scenario, 24 Mha (60 M acres) are planted with perennial grass, and the area under active cropland and cropland pasture each decrease by 10 Mha (25 M acres), compared to land use in yr2003.

The billion-ton vision report outlines factors that could act to decrease the recoverable supply of biomass. Among the factors related to forest biomass: accessibility (including environmental impact of gaining access), transport costs, and labor availability. For agricultural biomass, stated concerns include the realization of high yields, efficacy and economics of removing crop residues, and the market for biomass. Water supply constraints are not included as a factor in the report, nor are concerns regarding global food price, or impacts of land use changes brought on by increased demand for biomass.

It is important to note that crop residue is not necessarily a waste [62]. Wilhelm et al. show that sustaining soil organic carbon content to protect soil fertility and sustain high yields is the binding constraint which sets the limit on the fraction of crop residue that may be sustainably removed [63]. In addition, micronutrients are often bound with organic matter in soil, where they are protected from leaching and erosion. Thus, loss of soil C may involve loss of micronutrients, which may have to be added when (re)building soil C stocks.

A number of studies describe optimistic visions of the potential for biomass energy in the US. Gielen et al. [64] investigate the role of biomass in a GHG emission-constrained world. In this model, biomass supplies a substantial portion of transportation fuels and residential and industrial heating. Biomass does not provide much electricity, mainly because of alternative low-emission technologies. McLaughlin et al. [65] and Walsh et al. [66] report that farm gate prices of $44/ tonne DM of biomass energy crop would motivate the conversion of 9.4 M ha of cropland to switchgrass cultivation. They estimate 158-170 dry Tg of biomass energy crop would be harvested, and that biomass supply and crop prices would increase by 10-15%. This amount of biomass production could supply 3.1 EJ/ yr of primary energy, or 63.2 billion liters of ethanol (88.8 gal/ dry ton). Note that in yr2008, ethanol (EtOH) production from corn grain was about 34 billion liters.

Converse [67] outlines a renewable energy scenario that involves the use of 160 M ha of land for bioenergy production. In this scenario, 16 and 3 EJ of liquid and solid biomass fuel are produced, respectively. Converse notes that electricity generation using co-products of liquid biomass fuel production can reduce the requirements from other renewable electricity sources. Similarly, Lynd et al. argue that combined with increased energy use efficiency and altered energy use patterns, biomass could supply an important home-grown energy source [68].

As noted by Anex et al. [69], plant-derived material to be used for energy services need only contain C and H, and perhaps O (i.e. cellulose); other nutrients and compounds are seen in an energy context as unnecessary. Contrast this with food or fodder, for which other compounds (including protein and associated N) can be important sources of value.

Cellulosic biofuel feedstock may be harvestable from abandoned or degraded lands, and includes grassland perennials [58, 65]. Improvement in biomass-to-liquid fuel platforms could provide liquid fuel more cheaply than petroleum fuels, so GHG impacts should not be the only metric with which to judge the utility of biofuels [70, 71]. While refining bio-based products is currently more costly than for fossil refineries, opportunities exist for technological improvement and cost reduction [72]. On the other hand, the cost of hay in the US is over $100 per tonne (**Figure 1-7**), suggesting that the often-assumed biomass feedstock cost of $50 per tonne could be an underestimate. Furthermore, cellulosic EtOH production may compete with cattle for cellulosic biomass [73, 74].

Campbell et al. estimate that abandoned and degraded agricultural lands could supply 32-41 EJ of biomass per year globally [57] (6-8% of the world's primary energy consumption in yr2007). Such biomass grown for energy on marginal agricultural lands would not compete with food crops. This magnitude of biomass energy supply cannot provide a direct substitute for fossil fuels; the authors note "...the fact that the fossil fuel energy system already releases more C annually than that fixed by all croplands highlights the challenge of replacing a substantial part of the fossil fuel system with a system based on biomass." Keith provides a similar broad level perspective and asserts that accounting for the interactive effects of policy, land use and commodity prices is necessary for an assessment of biomass [75].

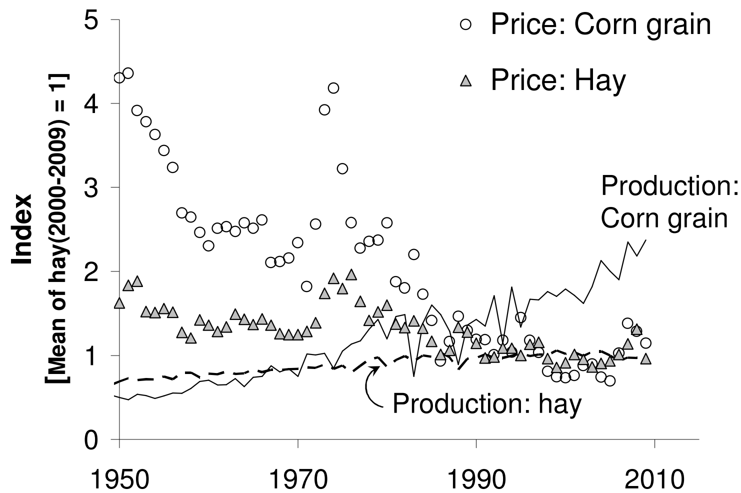

Figure 1-7. Price and production quantities of corn grain and hay (all hay varieties included). The points (dots and triangles) show price [$ / t dry matter (DM)]; both corn grain and hay price are indexed to the mean hay price, yrs 2000 through 2009. Similarly, production for corn grain and hay (DM harvested) are indexed to average hay production, yrs 2000 through 2009. Price and production data from [43]; prices are converted to constant dollars, yr2005, using CPI [76]. Around yr1970, corn grain and hay production quantities were similar; corn grain production has more than doubled since, while hay production has grown more modestly. Since ~yr1985, corn grain and hay prices [$/ t DM] have been comparable. The average price of hay over the years 2000-2009 was $130 / t DM, while average hay production was 119 Mt DM / yr.

1.5 Agriculture and climate: Review

Since humans already manage the majority of Earth's productive landscapes [77], and because of the strong interaction between the biosphere and global climate, there is substantial potential for climate manipulation through agriculture. Smith et al. report the physical potential for GHG abatement from agriculture worldwide is 5.5-6.0 Pg CO_2eq/ yr, with soil C management offering the majority of the abatement. The economic potential is 4.3 Pg CO_2eq/ yr with a CO_2 price of $100/ t CO_2 [78]. These potentials are subject to technical and policy constraints, and so the realizable total is likely much smaller [79].

Beach et al. use the DAYCENT [80] and DNDC [81] models to build GHG abatement curves (global and regional) for agricultural soil management, rice production and livestock management [82]. GHG abatement through soil management reductions typically come from more efficient fertilizer application, but the model also finds that at high GHG prices, less fertilizer is applied to reduce associated emissions, and yields are reduced.

Weber and Matthews report the climate impact of food (including production, transport, and marketing) to be 8.1 tonne CO_2eq/ household/ yr in the US [83], which amounts to 12% of US GWP in yr2005. The food system (including production, procurement, cooking, storage, etc.) has been estimated to use between 10 and 20% of technical energy in the US [84-86].

Reduced consumption of meat can also reduce GHG impacts from agriculture. Stehfest et al. develop dietary composition scenarios and use the IMAGE model to assess GHG abatement associated with the diet scenarios. The cost of stabilizing CO_2 concentrations at 450 ppm for the low-meat scenarios are about half of the cost with the "reference diet" scenario. Reduced land-use conversion and a terrestrial C sink from pasture allowed to revert to native vegetation offers large GHG abatement in the low-meat scenario [87]. Carlsson-Kanyama (with others) has calculated impacts of various food ingredients [88], and suggests that diet shifts away from ruminant animals toward plant-based food can offer GHG mitigation [89, 90]. Baroni et al. use the software tool SimaPro and also show that plant-based diets are environmentally preferred [91]. However, the use of air transport and extended refrigerated storage can increase the GHG impact of some plant-based produce to values close to those of meat [92].

Eshel and Martin calculate the energy and GHG impact of different diets, and highlight that the GHG impact difference between high and low-meat consumption diets is on par with the GHG difference of driving an SUV compared to an average sedan. They calculate that the GHG impact of the animal-product portion of the average American diet is 1.5 t CO_2eq/ person/ year. This impact includes enteric fermentation and nitrous oxide emission from manure management, and is calculated by allocating GHG emissions reported by the US EPA among various animal products, but it does not include N_2O from fodder production [93].

In addition to lower meat consumption, reducing food waste could also lower the overall impact of the food supply system. On a global level, food loss data are relatively scarce, but food waste tends to increase as food spending becomes a smaller percentage of income [94]. In the US, 15%-30% of meat, 23% of eggs, and about 20% of dairy products are wasted at the consumer level (including restaurant waste) [95].

1.5.1 Land use change

In the context of climate, if a large stock of relatively stable (and perhaps ancient) biomass is oxidized, this C added to the atmosphere could be considered a mobilization of a C store, rather than as being a part of the shorter-term C cycling. Old forests and mature grassland soils are classic examples. If these types of lands are converted to alternative uses with lower C stocks, the CO_2 emission from land use change (LUC) contributes to anthropogenic GHG additions to the atmosphere [96]. Land conversion in areas with large terrestrial biomass stocks has almost certainly been a significant contributor to atmospheric CO_2 enrichment by humans. These historic and current land use emissions are the consequence of a confluence of social, economic and moral considerations of the day, as well as of personal decisions and happenstance.

Humans have been altering land-based vegetation for millennia. There is evidence of forest management going back some 10k years [97]. In addition, agricultural development may have been responsible for atmospheric CO_2 concentration increases more than 5k years ago via land conversion and subsequent terrestrial C stock liquidation [98]. Olofsson and Hickler estimate that between 4000 B.C. (~6k years before present) and A.D. 1850, ~290 Pg CO_2 was released through land use change due to permanent agriculture, with another 120 Pg CO_2 released from non-permanent agriculture [99].

In the US, wood fuel consumption and agricultural plowing reduced forest and soil C stocks. Between the years 1700 and 1990, land use change in the US transferred 120 Pg CO_2 from the terrestrial C pool to the atmosphere. Forest and woodland area was reduced by 42%, and these lands accounted for 90% of C emissions [100]. Peak CO_2 emissions from LUC in the US may have been ~1.5 Pg CO_2/ yr around the year 1880 [100], or perhaps 2.9 Pg CO_2/ yr around year 1900 [101]. Currently, the terrestrial C pool in the US is increasing at a rate of 0.25 Pg CO_2/ yr; forest regrowth is responsible for most of this C sink [102].

Contemporary deforestation and forest degradation globally are estimated to have added between 2 and 10 Pg CO_2 to the atmosphere each year during the 1990s, while average annual CO_2 emissions from fossil fuel use and cement manufacture were ~24 Pg CO_2/ yr in the 1990s [103]. Notwithstanding forest regrowth and a probable undetermined terrestrial C sink [104-108], current rates of global C emissions to the atmosphere through human-induced forest destruction and degradation (mostly in the tropics, currently) may be unprecedented, although liquidation of the bioC pool in the US just before yr1900 was also substantial [101].

The amount of C stored in the soil and standing biomass varies greatly by ecosystem type, and the history of land used to produce a biomass feedstock is a key factor in assessing the life cycle C impact [109]. Tropical rainforests contain about 200 Mg C/ ha, and peat-rich rainforests contain over 900 Mg C/ ha [96]. In the US, land converted from row-cropped agriculture to perennial grass sequesters C at a rate between 0.0 and 3.0 Mg C/ ha/ yr [110-113] (for some of these higher values, fertilizer is applied). Total soil organic C content of soils under switchgrass was found to be ~15 Mg C/ ha higher than under crops [114].

The time period over which LUC emissions are allocated affects calculated GHG impact. Biofuel LCAs often use a simple uniform allocation over 20 or 30 years. However, since most emissions from LUC occur at the outset of production, a correction factor may be required to accurately account for emissions timing and atmospheric residence time. This correction may increase LUC GWP compared to uniform amortization [115, 116].

1.6 GHG balance of bioenergy feedstocks: Analysis

Biomass used as fuel is often considered "carbon neutral" or "climate neutral", meaning that burning biomass does not contribute to anthropogenic climate change. When biomass is combusted, CO_2 is released, but this CO_2 is often considered part of the short-term C-cycle, as the C in the fuel was (recently) extracted from the atmosphere. The actual GHG impact of a biofuel depends on the plants that would grow in the absence of biofuel production, and how they would be processed [117]. In addition, if harvesting biomass changes the stock of terrestrial C (above-or below-ground), this change should be accounted for. The C-stock change associated with a land-use practice may be depletion (as with deforestation) or enhancement (e.g. perennial crops). In this way, the GHG balance of biofuels depends on the history of management of the particular plot being harvested, and, critically, on the baseline stock of bioC assigned to the plot [118].

In addition, nitrous oxide (N_2O) emissions and fossil fuel use associated with producing crops affect the GHG balance of biomass. Contemporary agricultural practices rely heavily on fossil fuel-powered processes. Producing seed, preparing fields (where necessary), planting, producing fertilizer, harvesting and/or collecting, and processing the biomass usually release GHGs to the atmosphere because these production stages likely use fossil fuel. While these energy-using and carbon-emitting processes are important and should be accounted for, N_2O emissions often have a particularly large impact on the GHG balance [119].

The GHG emissions associated with processes in maize (corn) and switchgrass biomass production are shown in **Figure 1-8**. The total GHG impact of growing one kg of harvestable above-ground plant DM, not including soil C changes, is 260 and 130 g CO_2eq for corn and switchgrass, respectively. Note that this estimate includes some harvested corn stover. If only corn grain is harvested, the impact is 390 g CO_2eq / kg DM grain. These estimates agree with the ranges provided by Farrell et al. [120] and Kim et al. [121].

If we assume one kg of biomass DM corresponds to about 15 MJ of fuel input (after correcting for moisture and any processing requirements), and that one MJ of sub-bituminous coal fuel is associated with 100 g CO_2eq (life cycle), while one MJ of natural gas is associated with 60 g

CO_2eq [122, 123], using biomass to directly offset coal or natural gas (e.g. in a boiler for process heat or space conditioning) displaces about 1400 and 800 g CO_2eq, respectively, per kg of biomass DM.

While the estimates as shown in **Figure 1-8** are useful for visualizing relative impact of the processes, they require explanation and discussion of variability. First, consider that N_2O emissions are a large source of the GHGs associated with growing corn (**Figure 1-8**). N_2O emissions in agroecosystems are thought to be highest when soil is wet, e.g. during soil thaw [124], after rain [125], and with poorly drained plots [126]. Soils may also be a sink for N_2O, but this usually occurs when soil N concentrations are relatively low [127], which is not typical of agricultural fields. Emissions of N_2O are influenced by N source, application rate, climate, crop type and soil type [128-131], and so N_2O impact of growing biomass is variable [132], as reflected in **Figure 1-8** (see **Chapter 3, §3.3** for more detail regarding N and N_2O processes, measurement and modeling).

Next, consider that soil C accumulation under switchgrass, a perennial plant, has the potential to offset all other GHG emissions associated with growing the biomass. Soils converted from native prairie ecosystems to conventional cropped ecosystems lose a significant amount of C [133, 134]. When these lands are converted back to perennial grasses, soil C increases [110, 113]. The potential for "carbon-negative biofuels" was described by Tilman et al. [58]. Importantly, the degree to which a current agricultural practice will enhance soil C depends on past land management [111, 135]. The soil C content will eventually saturate, often within 30 years [135, 136], thus the period over which the soil C accumulation is amortized will affect the calculated per year accumulation.

Soil C under annual crops may be enhanced using "conservation tillage" practices [135], but these effects are uncertain. For example, when weeds can be managed with herbicide (made more feasible by the use of transgenic crops not affected by certain herbicides), conservation tillage allows farmers to prepare fields for planting by applying herbicide, instead of plowing. Soil C and N content affect microbe-generated N_2O, which may increase with conversion to reduced tillage [137, 138]. In addition, the finding that reduced till agriculture increases soil C may not hold if the soil is tested to sufficient depth [139-141].

Soil C accumulation under corn is assumed to be zero, reflecting a steady-state situation where well-managed annual cropping is the baseline. Since switchgrass is a perennial plant with potential to add more C to the soil than corn, the accumulation shown in **Figure 1-8** for switchgrass is 370 kg CO_2 / kg DM harvested (0.5 t C/ ha/ yr). Such a rate could be sustained for 20-30 years, and depends on the baseline soil conditions and regional climate (see **Table 1-2** for sources).

Consider this back-of-the-envelope calculation to illustrate potential LUC impacts: Suppose that some land under the Conservation Reserve Program[3] (CRP) in the US is converted to corn, and the conversion is associated with an emission of 10 Mg C from soil to the atmosphere. Considering a time frame of 30 years [142] and the corn DM yield in **Table 1-2**, the soil C loss

[3] The Conservation Reserve Program is a USDA program that pays landowners to convert environmentally sensitive cropland to long-term grass, trees or wetland.

associated with converting CRP land to corn amounts to 100 g CO_2eq/ kg DM of corn harvested from the converted land. This is slightly less than the impact of N_2O, and thus indicates the potentially important impact associated with land use change.

In addition to the direct impact of farming on soil C content, local land-use decisions can influence land-use globally through agricultural commodity prices. This indirect land-use effect can have a large influence on the life-cycle GHG emissions of biomass energy. See **§1.5.1** for more about C emissions from land use change and **§1.8** for discussion of the equilibrium effects of increasing biomass demand.

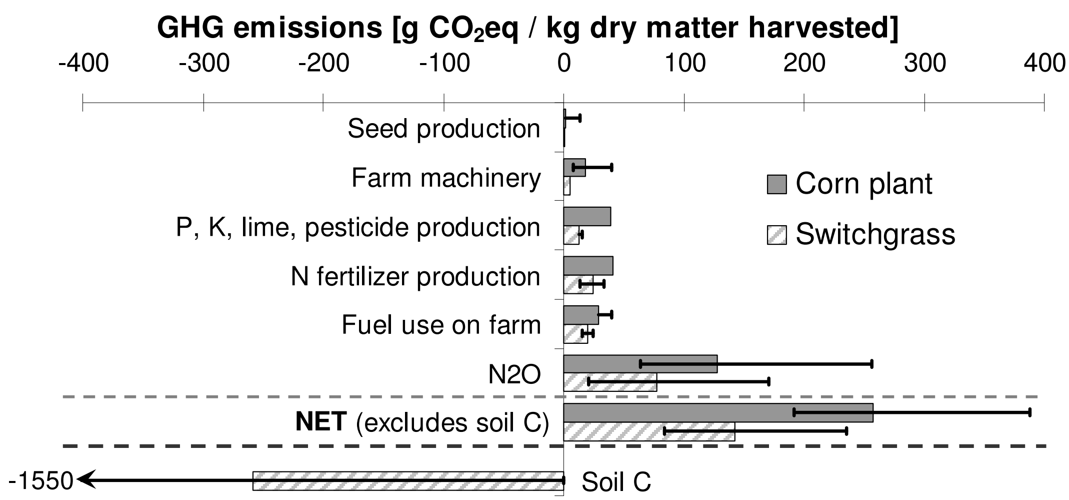

Figure 1-8. Comparing GWP (100 yr) of producing corn and switchgrass at the farm gate. "Corn plant" includes 50% of stover. Sources and assumptions are outlined in Table 1-2. A representative soil C accumulation is shown (not included in net emissions). Transporting 1 kg of biomass dry matter 250 km (50 km by truck, 200 km by rail) is associated with 35 g CO_2eq (assuming 20% moisture content for transported material; not shown).

Table 1-2. Parameters and assumptions for calculating GHG impact of producing land-based biomass. 'hDM' = harvested dry matter.

	Corn		Switchgrass	
Mg harvestable DM (hDM) / ha / yr	11.9	grain and 50% of stover harvested [121], representative of recent US grain yields at 150 bushel/ acre [43]; 1:1 stover DM:grain DM ratio at harvest [143]	7.1	[113, 144, 145]
N_2O emission: percent of N input	2% (1%-4%)	[121, 146, 147]	2% (1%-3%)	smaller range reflects potential for tighter N cycles with the perennial crop
t soil C emission/ ha / yr (as C)	0 (-0.1 - 0.5)	[148]	-0.5 (-3 - 0)	[110], range from [113, 149]
Farming inputs:				
m^2 land / kg hDM	0.8	(calculation)	1.4	(calculation)
kg seed / ha	20	[120]	2.3	[144]
g N / kg hDM	13.7	calculation using hDM and [150, 151]	8.3	[144]
g P / kg hDM	2.1		0.4	[152]
g K / kg hDM	4.7		9.1	
g $CaCO_3$ / kg hDM	39		na	
g pesticide / kg hDM	0.3	[150]	0.2	[144]
MJ fuel / kg hDM[a]	0.41	[150]	0.11 (±22%)	[144]
kg machinery & supplies / ha	35-55	[150, 153]	6.2	[144]; energy input per kg machinery from [154]
GHG impact of producing farming inputs:				
kg CO_2eq / kg seed [b]	1.5 (0.9 – 7.9)		1.1	[144], assuming energy has same GHG intensity of diesel
g CO_2eq / kg nutrient (N|P|K)	40.9 | 5.0 | 3.9	[150]	(same)	
g CO_2eq / kg ($CaCO_3$ | pesticide)	24.3 | 5.85	[150]	(same)	
kg CO_2eq / kg machinery & supplies (best estimate | range)	6.2 | 0.4-8.7	Best estimate using [154], assuming $10/kg farm equipment; [150, 153]	(same)	
Transport:				
g CO_2eq / tonne km by truck		180	[154-157]; described in [158]	
g CO_2eq / tonne km by rail		30		

[a] Fuel includes diesel, gasoline, propane, natural gas and electricity used on the farm

[b] For seed production, I apply Patzek's estimate that seed production is 7x more energy intensive than grain [153], and apply the non-seed GHGs calculated here. Range is from [159] and [153]. GHG intensity of energy used in production is assumed to be equal to that of diesel: 79.3 g CO_2eq / MJ [150]).

1.6.1 GHG balance of bio-derived fuels

In order for biomass to provide an energy service, it is usually processed. At a minimum, in order to use the feedstock as a solid fuel, it is transported, and usually chopped or ground, and perhaps pelletized. If the biomass is to be converted to a liquid fuel, processing may include hydrolysis (enzymatic, catalytic, or chemo), fermentation (and distillation), or gasification with catalytic recombination. Gaseous fuels can also be produced either chemically via gasification, or through biological anaerobic digestion.

In the literature on the GHG balance of liquid biofuels, N_2O emissions, ecosystem C changes (either through tillage or land use change), and credits for co-products are consistently major determinants of results [160, 161]. The fuel used to power the EtOH plant, as well as the age and design of the plant can also materially affect the GHG impact of EtOH [162].

Liquid fuel derived from cellulosic feedstock could have significantly lower GHG emission than from corn [58, 144, 163], and protein and nutrient recovery from biorefineries could reduce impacts further [69, 72, 164]. Perennial grasses allow continuous harvest with relatively few inputs, while supporting more diverse and robust sub-surface ecosystems than with annual crops [165]. The lower emissions associated with cultivation, the prospect for soil C enrichment, and the possibility for the cellulosic EtOH factory to be a net electricity exporter combine to provide the promise of low-C liquid fuel from biomass. Cellulose-based biorefineries are currently under development [166].

Liquid biofuels do not necessarily deliver the lowest GHG emission from biomass feedstock. Using a biomass feedstock to produce heat or electricity can offset more GHG emissions than using the biomass to make liquid biofuels [167, 168]. The cost of offsetting coal burning with biomass is consistently estimated at less than \$100/t CO_2 [51, 169-171]. If EtOH GHG emissions are 20% lower than those of gasoline, and if EtOH is \$0.50 more expensive than gasoline per gallon (energy equivalent basis), the cost of the GHG abatement is over \$300/t CO_2. Biomass processing improvements could reduce the cost of EtOH [68] and petroleum prices could increase faster than biomass prices (e.g. [70]), thereby making GHG abatement with biofuels cheap (or free). Alternatively, low-C electricity could power synthetic hydrocarbon manufacture using biomass as a C-source if liquid fuels are desired and low-C electricity is inexpensive [172]. Using biomass feedstocks to co-produce electricity and biochar could be an attractive mitigation options, with the magnitude of reductions dependent on the ratio of char and fuel production, type of energy offset, permanence of the stored char C, and N_2O and CH_4 soil emissions changes from char improvement [173]. Decentralized energy and charcoal production could also reduce GHG emissions [174].

Finally, GHG mitigation is not the only, or the primary reason liquid biofuel production is subsidized. Worries about being dependent on petroleum imports may trump worries about climate change. Achieving energy independence and reducing GHG through means other than biomass, such as C-neutral synthetic hydrocarbons [175], may be able to more efficiently address the need to reduce both GHGs and dependence on petroleum [176].

1.7 *Non-GHG environmental issues with bioenergy*

Even if corn-based EtOH does reduce GHG emissions, these climate impacts are not the only, or even the most important effects associated with growing plants for energy. Expanding corn-based agriculture to produce EtOH has other environmental impacts [177], including nutrient enrichment of ecosystems [178-180], reduced ecosystem services [181] and water consumption.

Irrigation in agriculture accounted for 34% of the total water use in the US in the yr2000 [182]. Continued expansion of grain-based biofuel production could exacerbate an already stressed water supply situation, particularly in the high plains states [183]. Chiu et al. report average embodied water in EtOH produced in the US as 140 L water/ L EtOH. However, the authors note that very large regional differences in embodied water make this average almost meaningless, since the range offered is between 5 and 2100 L H_2O/ L EtOH. For example, the states Nebraska, Kansas, Colorado and California together produced 17% of the EtOH in the US, but used 85% of the irrigated water required for EtOH feedstock production. The average embodied water was 672 L H_2O/ L EtOH in these states [184]. King and Webber compare water use in various transportation fuels, and report an average of 600 L H_2O/ L EtOH, with a range of 80 to 1500 [185]. Land under the CRP in Kansas was converted to corn, while Nebraska had record acres of irrigated corn in yr2008 [186]. If biofuel crops replace other, more water-intensive crops, water use could decrease [187]. Regardless, non-irrigated crop growth in the high plains is precipitation-limited, so achieving high yields from herbaceous crops in this relatively arid region will require water inputs. Increasing demand could also increase biomass prices and stress poorer households globally who already spend a high fraction of income on food [188, 189], although agricultural commodity price increases may cause labor wages to increase, which may benefit poor food-buyers [190].

1.8 Bioenergy, land use and climate change: Linkages

What services can biomass provide, and what are the environmental and economic costs and benefits of the various options? Biomass (whether a primary-, co-, or byproduct) can supply food for humans, feed for animals, energy, and materials. It also provides ecosystem services (e.g. material and energy cycling and perhaps as a climate moderator through C storage and respiration) and aesthetic amenities.

Being more self-sufficient in terms of energy certainly has benefits, but achieving this goal through increased biomass use also has negative repercussions (the potential to use under-utilized organic byproducts is reviewed in **Chapter 2**). Land managers make decisions regarding the product or service to which they will devote land. Such decisions are often based on financial criteria, over which government policies clearly have an influence. The motivation to support biomass-based energy resource use seems to rest on a three-legged stool. First, liquid fuels produced with "home-grown" biomass can offset some of the considerable petroleum imports into the US, thus improving domestic energy security [68, 191]. Second, biomass can be a low-GHG energy source [50, 120], although it is not necessarily so. Third, it can improve quality of life by stimulating economic development in rural areas and improving organic waste management. If biofuels provide value for society, their support via government mandate or financial subsidy may be justified.

However, policies that promote bioenergy can divert land use from conventional crops and grazing to bioenergy crops. Economic transfers from government to landowners and crop producers have affected land use in the US [192]. In addition, agricultural goods are transported by air, sea and land and are traded internationally. Early in 21st century, 13% of all crops (DM basis) are exported [193], including more than 20% of global oilseed production and 10% of coarse grain production the years 2004-2007 [194].

Thus, local market conditions are shaped by global economics. While global agricultural markets are not fully open, policies that promote and subsidize bioenergy influence agricultural practices and can lead to indirect (market-mediated) land use change (iLUC) – on this there is general agreement [195, 196].

The upward pressure on demand caused by biomass subsidies effects production levels and prices for other crops and the total cropped area [197]. Such changes in demand and price work to increase the amount of cropped land [198], and can provide incentive to convert uncultivated land to agriculture, thereby degrading ecosystems and reducing productivity [199]. Higher demand (and crop prices) can induce yield increases, but increased demand in the US via biofuel requirements also reduces coarse grain and soybean exports, which may then lead to increased crop areas globally [200]. The specific magnitude, location and timing of land use conversion induced by these biofuel policies is uncertain [201]. In addition, modeled land use change impacts of biofuel policy are critically dependent on assumptions regarding supply and substitution elasticities, as well as the ease with which imports and exports are adjusted [200]. Finally, the energy and agricultural sectors are becoming more integrated, and increasing fossil fuel prices can increase demand for biomass [202].

As discussed in **Section 1.5.1**, land use changes may be associated with large GHG releases. Searchinger et al. [203] model global cropland allocation, assuming that 112 billion L of EtOH are produced in the US in yr2016. Under such a scenario, 12.8 million ha of land in the US are diverted from feed production to ethanol production, while 10.8 million ha of land around the world (including Brazil and China) would be converted to cropland. Using a model of the US energy and agriculture systems, Johansson analyzes the competition for land between energy and food agriculture. Under the climate policy scenario, agricultural land rents are almost a factor of ten higher than in the baseline scenario. Similarly, wheat prices are almost a factor of 3 higher in the climate policy scenario [204]. McLaughlin et al. also find that expanded bioenergy increases food prices, but the magnitudes are more modest at 10-15% increases [65].

Kim et al. offer considerations which temper the utility of assigning impacts from iLUC caused by biofuel policy [148]. Perhaps the most poignant is the recognition that biofuel industries must not be held “responsible for the environmental consequences of decisions over which they have no control” [148] (p961). Thus, we might consider the policy itself as being responsible for emissions, given that iLUC emissions actually occur (note that biofuel policy is but one of various motivations for land use conversion [205]).

Kim et al. also argue that including biofuels in a GHG policy regime while excluding animal feeds (a large user of coarse grains worldwide) might not be “intellectually justified” [148] (p961). Biofuels do currently use a minority share of coarse grains. GHG abatement is one justification for supporting biofuel production, but mitigating climate change is not typically a motivation for feeding grains to animals (even if feeding animals may be influenced by biofuel and climate policy, particularly through coproduct feeding and GHG credit markets). Thus, it seems reasonable to estimate the iLUC impact of biofuels, while not addressing the GHG impact of the land use implications of animal feeding in the biofuel policy. These interactive effects continue to be an important area of study.

Unease over petroleum imports may drive biofuel policy, but biofuels are not necessarily the best way to use biomass for climate change mitigation. Righelato shows that reforestation is a more effective way to reduce CO_2 concentrations than producing biofuels, on a per hectare basis over 30 years [206]. Lubowski et al. find the cost of forest-based C sequestration to be roughly equal with the cost of C abatement with biofuels over a 250 year interval [207]. Similarly, Kirschbaum [208] discusses the tradeoff between using biomass to sequester C and provide fuel and finds that a biomass stand that is never harvested provides about the same climate benefit as a biomass plantation producing biofuel to replace fossil fuel, but does not discuss the possibility of harvesting biomass and sequestering the C in perpetuity. Pineiro et al. find that over a period of 40 years, setting aside agricultural land (as in the Conservation Reserve Program) provides more GHG offsets than would EtOH produced on the land by growing corn [142].

Schneider and McCarl [209] model bio-based fuel production in the US as a function of a carbon price. Their results indicate that biomass used for electricity generation provides most of the GHG abatement, and that cellulosic ethanol does not enter the market if the processing cost is above $0.50 per gallon. Carbon sequestration via afforestation and soil C enhancement dominates abatement up to $10 per tonne CO_2eq; biofuel starts to dominate after about $20 per CO_2eq. At $30, about 1 Gg CO_2eq is abated; at $150, about 1.5 Gg CO_2eq are abated and about 45 million ha of land are used to produce biofuel for power plants (note that about 35 million ha of land were planted with corn in yr2008; see also **Figure 1-2**).

Farmers are putting in place and refining practices to protect ecosystems outside the farm boundary. We may also develop improved pesticides with lower side effects and continue crop yield growth, while still improving the quality of the crops. Policies which insure farmers against sub-par grain yields can avoid over-application of fertilizer [210], and development of N-fixing crops could reduce impact further Perhaps most importantly, water use efficiency will likely also improve if water prices increase, and so more sustainable water use does not necessarily require reduced irrigation area.

While there is likely plenty of land on Earth with which to provide food to 10 billion people [33], the impact of agriculture on global ecosystems is major [211]. Policies that mandate or subsidize biomass energy production could increase demand for land and exacerbate the already large impact of the agricultural land-use system. Given that governments will act to reduce GHG emissions, the degree to which terrestrial bio-C (standing biomass and soil C) management is integrated into policy will have a large influence on land uses. If the C content of standing biomass and soils is not valued in climate policy, bioenergy crops could be the dominant land use globally [212], leading to the demise of unmanaged ecosystems, with the resulting impact on biodiversity, in addition to the emission of the bio-C stock at land conversion. At the same time, valuing terrestrial C will increase the cost of converting land to agriculture, possibly putting upward pressure on food prices. With regard to the development of large-scale bioenergy, Marland and Obersteiner are probably as optimistic as can be reasonably defended: "…biofuels can contribute importantly, but with thoughtful caution." [213]

The intention of the following chapters is to inform some of this cautious thought required regarding the development of biofuels. Given somewhat limited supply of terrestrial biomass, improving utilization efficiency and reducing waste is a useful activity (reviewed in **Chapter 2**). In addition, shifts in animal production can also have a profound impact on biomass demand, so animal production is reviewed in **Chapter 3**.

1.9 Thought experiment: increasing biomass supply

The US could increase domestic biomass supply (and perhaps improve energy security) by decreasing biomass exports. Another potential supply of biomass would result from animal product switching. Producing animal protein (or calories) with chicken, for example, is much less demanding of biomass than producing beef in the US (see **Chapter 3**), so the continued replacement of beef which chicken could make available a substantial amount of biomass feedstock. The scenarios outlined below are not meant as predictions, but are presented to illustrate potential trade-offs. Appropriating all biomass exports for biofuels would likely have systemic effects on crop prices and cropping practices globally, and so these scenarios should not be viewed as promoting biofuel expansion.

Now, consider crop exports. As shown in **Figure 1-9**, relatively large fractions of corn grain, soybean, and wheat production in the US is exported. If the US were to aggressively pursue "home-grown" energy, the land used to produce exports of corn, wheat, soy and hay could instead be used to produce biofuel. These crop exports currently use 31 M ha of land (**Table 1-3**). If we assume this land can produce perennial biomass at 7 t DM/ ha/ yr, appropriating exports could supply ~220 Mt DM / yr.

Of course, decreased exports of biomass and associated higher prices in global markets could have negative consequences, with the possibility of increased malnutrition being one of the most important. What if 20k people die each year from the biofuels mandate through reduced access to food? Assuming a life value of 4 M$, and the biofuel production is 32 G gal, the cost of deaths caused by malnutrition would be $2.50 / gallon. Writing in such callous terms about value of lives lost to malnutrition is somewhat absurd, but the possibility that biofuel mandates could increase food insecurity is not. If people who are food stressed are farmers producing agricultural commodities, increased prices could be a net benefit. However, for those who use a large share of wages to purchase food, increased prices are clearly a negative.

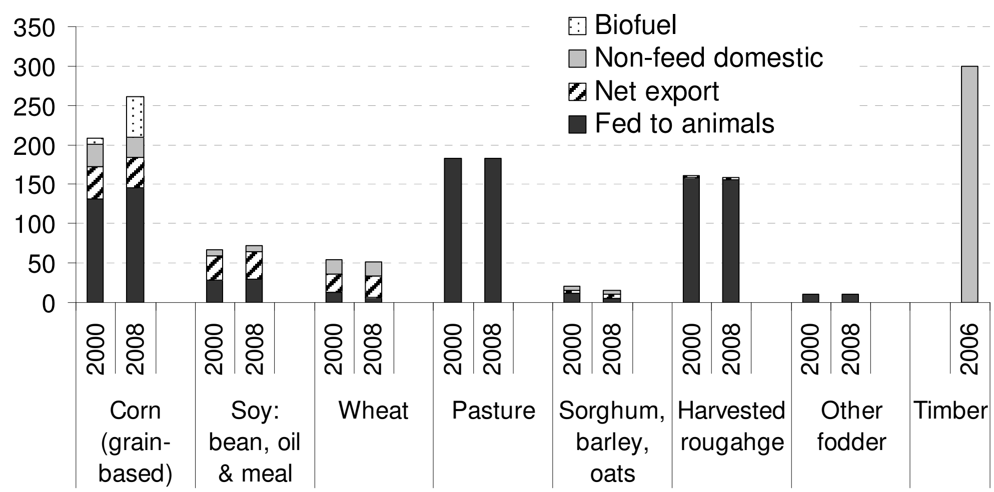

Figure 1-9. Production and utilization of major crops in the US; crop data from [43, 214]. Harvested biomass from forestry ("Timber") is shown for scale [215] (see text for calculation details).

Table 1-3. Calculating land used in US to produce crops for net export (export minus import). Net exports and yields are averages of years 2005-2009, with standard deviations (StDev) shown.

	Net exports		Yield		**Land**
	[Mt DM]	(StDev)	[t DM / ha]	(StDev)	[M ha]
Corn	44	6	8.1	0.3	5.4
Soybeans+	32	4	2.5	0.1	13.0
Wheat	28	7	2.4	0.2	11.4
Other grains	3	1	2.8	0.7	1.1
Hay	1	0	4.9	1.3	0.1
TOTAL	108	10	3.5	1.6	**31.0**

NOTE: "Soybeans+" includes whole soybeans as well as soymeal and soy oil.
Sources: Trade data for grains from [214], soybeans from [216], hay from [217]; Yield data from [43].

Another way to increase the available supply of biomass for energy would be to replace all beef production with an alternative, less fodder-intensive animal product (on a per edible calorie basis). In yr2007, beef cattle consumed 38 Mt DM of concentrate feed (including but not limited to corn and soybeans) and 198 M t DM of roughage feed (including hay, green chop, and grazed biomass). To replace the 22 T kcal per year supply of beef calories with chicken (or some other animal which has similar feed conversion to food calorie efficiency) would require 40 Mt DM of concentrate feed. Thus, replacing beef with chicken would liberate ~200 Mt DM of roughage. This could be increased to 300 Mt DM or more if biomass feedstock could be managed to produce higher yields.

Thus, the annual supply from appropriating crop exports is 220 Mt DM and the supply from animal switching is 200 Mt DM / yr. With an energy content of 19 MJ (HHV) per kg DM, these biomass supplies together amount to about 8 EJ per year.

Wood harvested from trees is also a large source of biomass feedstock in the US (**Figure 1-9**). About 300 Mt DM of tree matter was harvested in yr2006 (calculated from harvested volume in [39], assuming 0.6 t DM/ m^3 softwood and 0.85 t DM/ m^3 for hardwood [215]). The rate of wood harvested amounts to about 60% of the total forest growth [39]. Thus, the supply of forestry biomass could be increased without necessarily reducing the forest stock. The National Academy of Engineering estimates sustainable utilization at 110 Mt/ yr in addition to current use [218], while the estimate from a USDA/DOE report gives 330 Mt/ yr (including collecting material currently left on the ground during tree harvesting, e.g. bark) [48].

1.10 Appendix 1: GHG background

"Global Warming Potential (GWP) is a measure of how much a given mass of greenhouse gas is estimated to contribute to global warming. It is a relative scale which compares the gas in question to that of the same mass of carbon dioxide (whose GWP is by convention equal to 1). A GWP is calculated over a specific time interval and this time interval must be stated whenever a GWP is quoted or else the value is meaningless." (quoted from [219]; (see also [220]). The 100-year GWP of methane and nitrous oxide are 25 and 298, respectively (the 500-year GWPs are 7.6 and 153, respectively) [220]. Thus, for example, an emission pulse of 1 g of CH_4 would cause the same warming over a period of 100 years as an emission pulse of 25 g CO_2.

Coal cost per GJ is calculated using price [$/short ton] and heat content [Mbtu/short ton] from [49]. CO_2 intensity of US coal is 90 g CO_2/MJ (converted from 26 Tg C/quadrillion btu), from [221]. These emissions are only from burning the fuel, and do not include emissions from mining, processing and transport operations and associated infrastructure and capital equipment.

Table 1-4. Estimates of global CO_2 emissions from fossil fuel combustion and cement manufacture. Units= [Pg CO_2 / yr].

Year	(average of →)	CDIAC [222]	Netherlands environmental assessment agency [16]	IPCC [103] (Table 7.1)
1990	22.2	22.5	20.6	23.5
1995	22.6	23.4	21.0	23.5
2000	24.5	24.7	22.3	26.4
2001	24.7	25.3	22.4	26.4
2002	24.9	25.5	22.8	26.4
2003	25.7	26.7	23.9	26.4
2004	26.5	28.1	25.0	26.4
2005	27.2	29.2	25.9	26.4
2006		30.2	26.8	
2007			27.6	

Table 1-5 Global non-CO_2 GHG emissions [223]. Units= [Pg CO2eq/ yr], using GWP of 25 for methane (CH_4) and 298 for nitrous oxide (N_2O); 100 year time horizon, from [220] (Table 2.14).

Year	CH_4	N_2O	high GHG
2000	7.2	3.0	0.4
2005	7.6	3.1	0.5

Table 1-6. CO_2 emissions associated with land use (net carbon transferred from terrestrial stocks to the atmosphere). From IPCC [103] (Table 7.2). Note the large range, indicating substantial uncertainty.

Year	best estimate	low	high
1990's	5.9	1.8	9.9

Table 1-7. Summary of global GHG emissions estimates. "CO_2 from fossil" includes emission from cement, and is the average of estimates in Table 1-4. GWP is reported for 100 year time horizon. Units= [Pg CO_2eq/ yr]

	Total	CO_2 from fossil	CH_4	N_2O	High GWP	Land Use (CO_2)
1990		22.2				
⋮						
1995		22.6				
⋮						
2000	40.9	24.5	7.2	3.0	0.4	5.9
2001		24.7				
2002		24.9				
2003		25.7				
2004		26.5				
2005	44.3	27.2	7.6	3.1	0.5	5.9

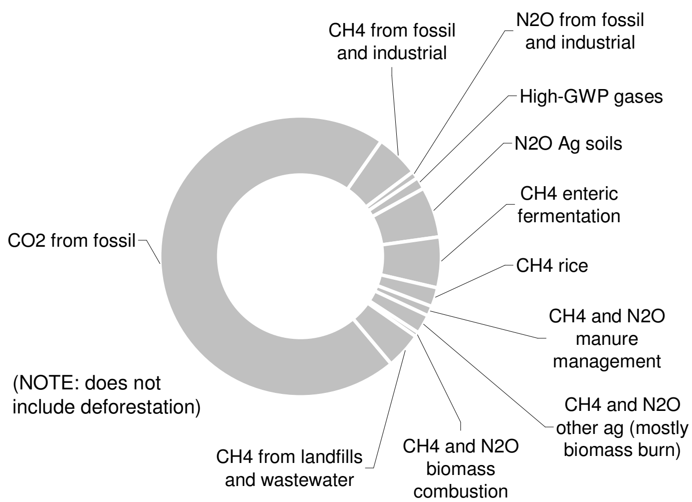

Figure 1-10. Global GHG emissions (CO_2 equivalent, 100-year timeframe) in yr2005; data from US EPA [223]. CO_2=carbon dioxide, CH_4=methane, N_2O=nitrous oxide; "Ag"=agriculture; "LUC"=land use change.

2 Energy from biomass byproducts

Abstract. This chapter first offers a review of estimates of the organic byproduct supply in the US. Byproducts with positive economic values are often considered co-products; when the byproduct does not offset production in any other system, it may be considered a waste. Utilization of a biomass byproduct for energy service provision avoids increasing demand for land to the extent that the biomass byproduct would have been otherwise unused. Annual byproduct biomass production of 500 Mt dry matter could supply 7.5 EJ of energy in the US (roughly 1/3 the magnitude of recent domestic coal utilization); currently, about 115 Mt are utilized. The thermal energy contained in municipal organic wastes amounts to 3 EJ (about 50 watts per capita). Crop residues could potentially provide twice this amount, but such residues improve soil fertility, so removal is likely to involve added costs (via increased fertilizer or water requirements or reduced yields). A simple framework is presented and used to perform a life cycle assessment of municipal organic waste management systems. The GHG impact of landfill disposal, anaerobic digestion (AD), and composting is compared. Probability distributions were used to characterize important processes, including methane (CH_4) collection efficiency and the fraction of C sequestered for each system. Two landfill cases are presented to account for the possibility of uncontrolled CH_4 emissions. The landfill (with high CH_4 collection efficiency), AD and compost scenarios all manage waste with roughly equal net GHG emissions (55-115 kg CO_2eq/ t waste), while the landfill (leaky) scenario emits about 400 kg CO_2eq per tonne of waste managed. Dealing with high-moisture, nutrient-rich residues from anaerobic digestion could provide challenges for organic waste managers, but the solid fraction could also supply revenue as soil amendment. Composting also provides a quality soil amendment (assuming it is not contaminated with metals), and so is an attractive organic waste management technique. The cost of landfilling and the demand for compost will determine the fate of waste organic material in the future. Increasing demand for (or further subsidization of) biofuel feedstocks could shift some organic material from the category of waste to co-product.

2.1 Introduction

A byproduct is often described as what it is not: it is not the primary output from a process; it may have a positive or negative value. A negative-valued byproduct is often referred to as waste. Using the perspective of the consequential LCA (discussed in **Chapter 3, §3.4**), waste is an output that does not displace any production in other systems [224]. A byproduct with positive value is often considered a co-product. Distinctions between by- and co-products shift as other influences (e.g. market and technological) change, and so these two terms must be viewed with some flexibility.

Agriculture and food-related activities are often associated with by- and co-products. These organic byproducts can be converted to EtOH [225, 226], used as animal feed [227], converted to biogas (and perhaps used as vehicle fuel [228]), incinerated to recover energy and reduce waste volume [229], pyrolyzed to harness energy and produce biochar [173], or be used as a material feedstock [230]. Even crop residues which are not harvested (e.g. corn stover left on the field) may be co-products, since they provide functions like erosion control and nutrient cycling, and perhaps soil C sequestration. This value may or may not be marketable and is partially dependent on the perspective taken by a land manager.

Through all this, we must remember that a waste in one circumstance is a resource in another. If I keep a pig (either for company or food) and I have food scraps, those food scraps might be valuable to me as pig feed. Likewise, food scraps might be valuable to a gardener who uses compost to maintain soil quality, while to another person, food scraps are garbage. A plastic bottle that is worthless to a household may have value at the gate of a plastics processing plant. Thus, the term ‘byproduct’ is adopted here to include both wastes and materials that have value.

Costs associated with labor (time) and transport distance influence the value of byproducts. Since individuals and organizations have differing cost structures and access to markets, it is not surprising that "win-win" situations exist where a "waste" generator and a "recycler" each benefit – the waste generator is rid of an unwanted material, and the recycler gets a valuable feedstock. As the conditions of this arbitrage change, the waste generator may realize the value of the material for herself – thus, the waste becomes a co-product at the point of generation (a byproduct with positive value). More generally, as supply and demand situations change in the byproduct and related markets, the economic nature of the byproduct collection could change from a revenue-generating waste collection to a cost-imposing feedstock acquisition. It is important to remember this dynamic potential when classifying byproducts as wastes or co-products, and when considering the economics of biomass processing.

This chapter provides a foundation for understanding the quantity of organic byproduct flows in the US. In addition, a framework for determining the GHG impact of using organic byproducts is presented and applied to municipal organic byproduct management. Probability distributions are used to characterize important processes in the management system.

2.2 Byproduct production and management

Waste management is an idea that encompasses many of the processes involved in dealing with the byproducts of living (and producing). It seems the desire for an environment free from rankness is one that many of us share with our ancestors. Liquid sewage and other byproduct streams (often rich in organic matter) are handled with on-site or centralized wastewater treatment facilities. Solid waste is often classified into municipal solid waste (MSW) and industrial, construction & demolition waste (ICDW). Waste management strategies may be very simple - indeed, humans have used garbage pits for millennia. Other simple strategies include piping sewage to a river, and piling manure removed from feedlots. With larger human populations and more concentrated animal feeding, such minimal byproduct processing has largely become unacceptable. This basic idea has led to waste water treatment for human and industrial wastes, engineered sanitary landfills for solid wastes, and, increasingly, manure treatment on large animal farms (see **Chapter 5**).

The magnitude of municipally-sourced byproduct generation (which includes material discarded and recycled) is uncertain - the two major estimates differ by 70%. The US EPA reports MSW production at 0.74 t a-c/ p/ yr (tonne as-collected mass per person per yr) in yr2008 [231]. The other estimate, which is based on a state-level survey conducted by BioCycle, reports MSW production at 1.25 t a-c/ p/ yr in yr2006 [232]. Both these estimates include recycled and composted material, but they do not include ICDW.

The majority of MSW generated is landfilled. The BioCycle survey estimates that 64% of MSW was landfilled, while 29% was recycled and 7% was incinerated in yr2006 [232]. The EPA estimates 54% of MSW was landfilled, 33% was recycled and 13% was incinerated in yr2008 [231]. Generation and recovery of MSW (as reported by the EPA) are shown in **Figure 2-1**, grouped into six categories. The recovery rate of paper and paperboard is relatively high (55%). Likewise, about 65% of yard trimmings are recovered (composted), but the recovery rate of wood and food scraps in the MSW stream is low. It is worth noting that 68% of the total MSW stream is organic in nature (63% when dry-matter corrected, see **Table 2-2**), including paper and paperboard, textiles, wood, food scraps and yard trimmings. About 40% of these organics are recovered, and they made up 80% of all recovered MSW in yr2008 [231].

Unfortunately, data on other solid wastes, including ICDW, is sparser than for MSW. The EPA estimates construction and demolition (C&D) byproduct at 0.5 t/ p in yr2003, and reports that for states with available data, 48% of C&D debris is recovered [233].

In addition to the rank and un-hygienic nature of organic waste, many wastes are associated with toxicity or ecosystem-disrupting potential. Thus, modern waste management involves managing and limiting interaction between the toxic components of human byproducts with ecosystems and humans. In the US, landfills must be designed with lining, leachate treatment and monitoring, and gas collection. Moisture content and flow has a large effect on biological and chemical processes within the landfill. Modern waste incinerators have extensive emission control, and pollutant concentrations in air and soil around such facilities are often indistinguishable from the ambient environment [234]. However, up to 10% of incinerator input mass must be landfilled, and air pollution control residues are hazardous waste and must be handled carefully (e.g. [235]).

Energy harvesting from garbage can be thought of as part of a material cascade [236, 237]. If a product like wood can be used more than once, it reduces the forest resource necessary to supply services [238]. For example, wood destined for the landfill could be pulped to make paper; this paper could be used first as white paper, then again as newsprint or paper board; it could finally be used for energy in an incinerator.

While proper handling of wastes is necessary to ensure adequate sanitation, creating markets for such byproducts can increase the value of byproducts. This, in turn, can subsidize consumption of the original product by reducing the cost (or increasing the revenue) associated with the byproduct. The cost of waste management is small, however, relative to other production-related expenditures, and waste management is often driven as much by aesthetic and hygienic concerns as costs. Thus, the rebound effect of increased consumption due to increasing byproduct value is not considered further.

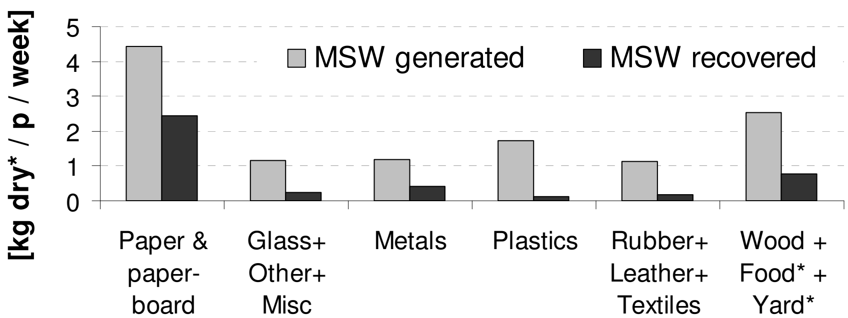

Figure 2-1. Municipal solid waste (i.e. byproducts) generated and recovered (i.e. recycled) in the US for yr2008. Units= [kg dry mass/ person/ yr]. These data are from [231], with some categories combined to facilitate interpretation (Note the BioCycle total estimate is 1.7 times higher); they do not include industrial and construction and demolition debris. *Food waste and *yard trimmings are based on 0.30 and 0.55 kg DM / kg fresh, respectively (see Table 2-2).

2.2.1 Magnitude of organic byproduct available in US

Organic material in the MSW stream is a small part of the total production of biomass byproducts in the US. Various estimates of potential biomass byproduct supply are presented in **Table 2-1** (the table also includes estimates of energy crop potentials for comparison). Gronowska et al. also review biomass supply estimates and finds a large range [239]. Energy content of biomass is often reported as 20 MJ/ kg DM. However, I will apply 15 MJ/ kg DM here to account for sometimes high water content in biomass residues, as well as transport and conversion inefficiencies. For the US, a total biomass supply of 500 Mt/ yr is at the upper range of a

National Academies (NAS) report, and could supply 7.5 EJ per year – this value includes purpose-grown bioenergy crops. The energy content of wood, food and yard waste in the MSW stream is about 0.5 EJ/ yr (50 watts per person), while the energy content of non-recycled paper products is about 0.6 EJ/ yr (60 watts per person). In order to put these values in perspective, about 24 EJ of coal were used in the US, and about 23 EJ of petroleum was imported in yr2008 while the total primary energy consumption was 105 EJ. Thus, the estimates indicate that the total biomass supply could supply a modest portion of energy services demanded (i.e. 7.5 / 105 = 7% in the US). Regionally, the percentages are likely non-uniform owing to differences in population density and agricultural productivity. On a per-capita basis, the energy available annually from biomass in the US (estimated at 7.5 EJ / yr) amounts to 800 W (heat content) per person.

Crop residues are a substantial part of the estimated byproduct biomass supply - corn stover in particular has been shown to be an abundant potential biomass source [23]. However, crop residues are not necessarily a waste [62]. Removing residues can influence yields [240] and soil C content [63]. Agronomic impacts of stover removal are site-dependent, however, and have been difficult to distinguish from weather-related influences [241].

Organic byproduct management is associated with relatively small portions of the total GWP (100yr) of human actions. In the US, emissions from organic residue management include CH_4 and N_2O from landfills, wastewater treatment, manure management, and other agricultural sources including field burning. These emissions of CH_4 and N_2O from organic materials account for about 3% of GWP in the US and around 7% globally, not including C emissions from land use change (**Figure 2-2**). Post-consumer waste and wastewater treatment make up about 3% of global GHG emissions. Bogner et al. estimate that GHGs priced at $100/ t CO_2 would provide incentive to mitigate 70% of these emissions [242]. Landfills emitted 22% of CH_4 emissions in yr2008 in the US [102]; the energy contained in these emissions amounted to 0.3 EJ.

Table 2-1. Various estimates of organic material available for human use in the US. Values shown are [Mt DM/ yr]. Most sources report a range of values, which are shown here. (see Appendix for discussion).

	NAS [218]	DOE/ USDA [48]	Tilman [243]
Dedicated crops & bioproducts [a]	90 – 150	15 – 480	75 – 90
Crop residues and waste (including manure)	100 – 145	160 – 425	90 – 135
Wood residues	100 – 110	250 – 330	65 – 130
Paper, food, yard trimmings, etc	60 – 65		20 – 35

[a] "DOE/USDA" includes 14 and 122 Mt DM of grain for the low, and high estimates, respectively

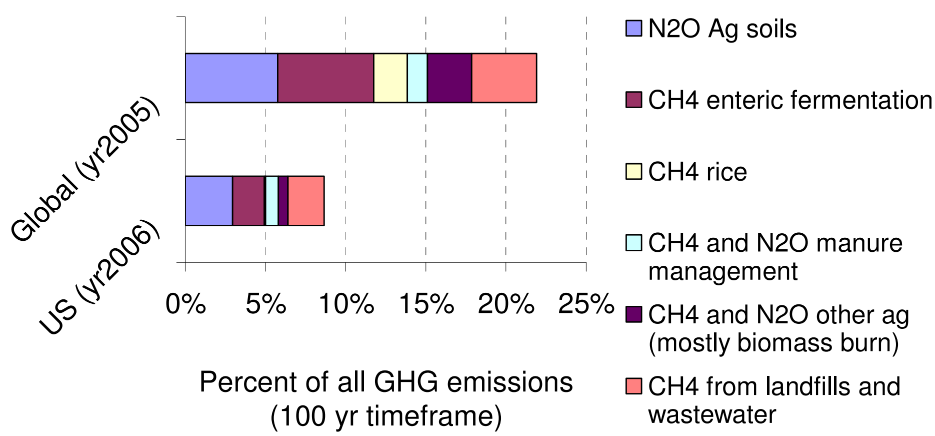

Figure 2-2. Non-industrial GHG emissions in the US and globally. Emissions from various source classes are shown relative to the total annual GWP of GHG emissions, using 100 year GWP factors. Emission data from US EPA [223, 244] and do not include emissions from land use change.

2.2.2 Managing byproducts: Literature review

There is no shortage of analyses which deal with municipal waste, including organic residues, and a variety of methods are available for judging the desirability of various waste management schemes [245]. Many assess the MSW management system, comparing landfilling, combustion, energy recovery, and composting. Some consider waste as an agricultural input, either as fertilizer or fodder. In **Chapter 5**, I present a detailed life cycle assessment of energy capture from animal manure. Here, I review studies of municipal waste management, with a focus on GHG emissions.

Source separation can help ensure safe use of MSW compost in agriculture [246]. Nutrients in organic household and municipal wastes and wastewaters are often dilute, so transport to fields is not an option. However, the shift from nutrient removal to nutrient recovery could improve nutrient use efficiency through concentration and redistribution [247].

While trash incineration with energy recovery is environmentally superior to landfilling and using fossil fuels, recycling metals and plastic materials reduces GHG emissions [248]. Recycling glass may reduce emissions slightly, but is highly dependent on collection practices, and is not likely to be economically attractive [249]. Recycling paper also may reduce emissions associated with deforestation for virgin paper manufacture.

Changes in handling and utilization of waste (both solid and liquid) in Aalborg, Denmark eliminated 200 kg CO_2eq/ p/ yr. Landfilling was ended (along with subsequent CH_4 emissions), MSW was incinerated (producing emission offsets from electricity and heat) and sewage sludge is digested anaerobically, among other changes [250].

Chen [251] models MSW management and GHG impact in Taiwan, and finds recycling the most effective way to reduce emissions. Diaz and Warith developed the WASTED model, which reports emissions inventories for various waste processing systems, including landfilling, recycling, composting and energy recovery [252]. Carlsson Reich offers an economic assessment of waste management systems, merging financial and environmental life cycles to determine the welfare economics of the systems [253].

Wilsenach et al. report that source separation is a promising method to aid in resource recovery from waste water [254]. Muchovej and Pacovsky outline how wastes can be used as valuable agricultural inputs. The authors note that the majority of literature finds that the benefits of using waste in agricultural outweigh the harm. However, the authors seem to understate potential risks [255].

The LCA-IWM is a model developed by den Boer et al. to assess the sustainability of waste management systems [256]. Cherubini et al. assess residue management in Rome. Of the fraction of residue that is waste (that is, not recycled or separated), the authors find that landfilling performs relatively well for the material flow indicators (particularly for water). However, the alternatives of anaerobic digestion (AD) and incineration are superior for net energy, GWP and ecological footprint [257].

Eriksson, Sonesson and colleagues use ORWARE to evaluate various solid waste management alternatives, including landfilling, plastic recycling, incineration, and AD. The landfilling scenario has highest GHG emissions and the plastic recycling scenario has lowest GHG emissions [258, 259]. Finnveden et al. compare management of particular fractions of MSW in Sweden and find recycling with AD to be the lowest GHG option for newspaper and plastics management. However, these results may be affected if personal vehicles are used to transport materials for recycling [260, 261]. Kirkeby uses the EASEWASTE model for Denmark to calculate the environmental impacts of solid waste management, comparing AD to incineration. Both options reduce GHG emissions, although arsenic present in organic waste poses a health risk when digested matter is spread on fields [262]. Diggelman and Ham compare managing food waste as either a solid or as liquid wastewater. Composting food waste as a solid material has lowest GHG emissions (nitrous oxide (N_2O) is not considered) [263]. Lundie and Peters perform an LCA of food waste management. Home composting that is properly managed performs best in terms of GHG, followed by wastewater disposal. However, landfill gas is flared without energy recovery, and N_2O emissions are not included [264]. Wilson compares the MSW management in Pamplona, Spain to various scenarios and finds that composting the organic fraction would lower GHG emissions by about 25% [265].

Ekvall et al. provide a balanced discussion of the limitations of using the LCA method for waste management environmental assessments. They also present ideas for addressing these shortcomings [266]. Some limitations include: the fact that the effects of waste reduction measures are difficult to account for; aggregation (of emissions) looses information which may be important; waste management system impacts and costs can be lumpy (owing to logistics and material processing facility investments), making linear models inaccurate; and static modeling of waste production rates. After calls to dynamic modeling of waste generation, geographic and spatial data cataloging, and using non-linear models of waste management (p991,992,993), the authors remind the reader that LCA models likely don’t account for all important aspects, and that LCA results should be viewed with an appreciation for quantitative uncertainty about a model’s parameters and equations.

2.3 Organic byproduct management assessment: A simple model

In the simplified GHG assessment presented here, the organic byproduct management (OBM) system is modeled by estimating impacts and offsets associated with five steps:

1. **Feedstock acquisition**
2. **Transport** from point of generation to point of processing, and from processing facility to point of use or storage
3. **Processing** the byproduct
4. **Use** of the feedstock or products from processing
5. **Storage** and sequestration of biomass C

In this section, the GHG impacts of four OBM systems are estimated using this framework. These steps are intended to represent a wide variety of possible actions. Collection, for example, might entail truck, barge and train transportation of a byproduct from a restaurant in NYC to a processing facility, or it could be simply walking from the kitchen to a compost heap to deposit material. The systems considered are:

1. **Landfill (baseline)**: Methane collection efficiency of a sealed landfill cell is representative of the overall methane collection efficiency
2. **Landfill Leaky**: A landfill where methane from easily degradable organic matter (e.g. food waste) is emitted before the methane collection is initiated
3. **Anaerobic digestion**: Organic material is digested anaerobically
4. **Composting**

A generic organic material is used to represent the organic byproducts available for collection. The properties shown in **Table 2-2** reflect a mix where 35% of the DM is food byproduct and 65% is yard byproduct (grass, leaves, and woody clippings, and such material from public areas). The C content of organic material is assumed to be 50% of the non-ash DM. **Figure 2-3** shows the distribution assigned to represent ultimate methane yield of the organic material (B_0, the maximum amount of CH_4 that can be produced from a feedstock).

Table 2-2. Properties of the organic mix, and food residue and yard-sourced portions. Material properties are from [10, 263, 267-270]. The ratio of food:yard byproducts reflects waste generation as reported by EPA [231]. See appendix for details of yard-sourced organics.

		Organic byproduct mix	Food-sourced	Yard-sourced
Dry matter	% of as-collected weight	43%	30%	55%
	% of DM in mix		35%	65%
Volatile solids	% of DM	88%	91%	86%
	% of VS in mix		36%	64%
Ultimate CH_4 yield (B_0)	L CH4 / kg VS	230	380	150
Carbon	% of DM	50%	50%	50%
	% of C in mix		35%	65%
Nitrogen	kg N / kg DM	0.018	0.03	0.01
Phosphorus	kg P / kg DM	0.003	0.01	0.00

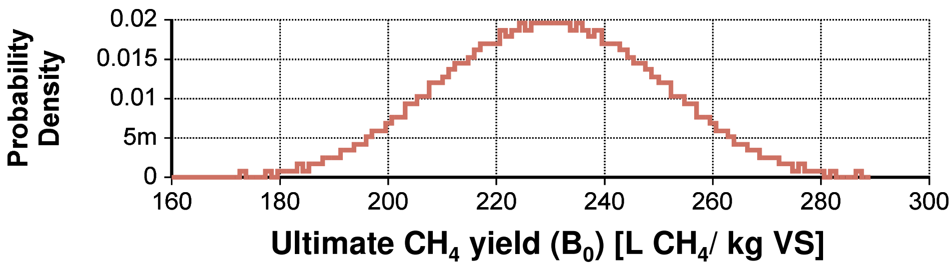

Figure 2-3. Probability distribution assigned for the ultimate methane yield (B_0 with units [liter CH_4/ kg VS]) of the generic organic mix. Adapted from CH_4 yields of waste components in [267-271].

2.3.1 Acquisition

If an organic byproduct is truly a waste, it does not carry any economic value, and it does not offset any other production [224]. In this case, it is appropriate to assume there are no upstream impacts associated with the byproduct. If, on the other hand, the byproduct does have some value, impacts associated with its production should be estimated using standard LCA procedures.

Available utilization options could affect the value of byproducts, and thus affect production levels of the original product. Integrated models and increased model specificity can increase the accuracy of analyses by accounting for dynamic interactions between waste generation and cost (or value) of options [266]. However, the aim here is to understand the source of major impacts, and broad level benefits of a few OMB systems, and so such equilibrium economic models are not employed. The organic byproduct management systems considered here are assumed not to increase the demand for biomass. Thus, no land use impacts are associated with the feedstock acquisition. Elaborations on feedstock acquisition are presented in the **Appendix**, **§2.5.3**.

2.3.2 Collection

For this assessment, the GHG emissions associated with collecting the organic material are assumed to be 44 kg CO_2eq/ tonne as-collected (t a-c) of material [263]. This impact is representative of curbside collection in a municipal environment. Emissions might be lower or higher depending on collection density and transport distance [252, 272].

2.3.3 Processing

The methods used to process biomass are diverse and situation-dependent. When biomass is to be placed in a landfill, it might be shredded and compacted. For compost, the material might be periodically turned, or mechanically aerated. For energy recovery with anaerobic digestion (AD), water might be added and the material might be pulped before entering a reactor vessel. Engineered landfills include a lined landfill "bottom", leachate circulation and treatment (leachate refers to the liquid that collects at the bottom of a landfill cell), monitoring of surrounding water for contamination, and gas collection to reduce CH_4 and VOC emissions. In addition to emissions from fuel use and equipment manufacture, biomass processing is often associated with CH_4 and N_2O production. Composting emissions of CH_4 and N_2O are sensitive to management and elemental makeup of the material. The net GHG emission from a landfill organic management system is very sensitive to the efficiency of CH_4 collection.

For compost operations, the energy (including embodied in equipment) required for processing is about 500 MJ/ tonne waste [263]. Diesel fuel (with life cycle GHG intensity of 79 g CO_2eq/ MJ) is assumed to power material processing at landfills and compost operations; the processing thus emits about 50 kg CO_2eq/ tonne, when considering equipment and fuel. In this analysis, it is assumed that the processing step for landfill and AD operations are twice as intensive per tonne of material processed than for composting.

Methane emissions from landfills can occur over decades as the material in the landfill is degraded anaerobically. Maximum CH_4 production typically occurs within 2 years after material is placed [268, 273]. Emission rates are particularly sensitive to the moisture content within the landfill. Landfills are required to install gas collection equipment to limit emissions to the atmosphere.

Methane yield obtained in the landfill is assumed to be 70% of the ultimate methane yield (B_0), while the AD achieves 95% of B_0 (**Figure 2-4**). Methane emissions from composting are estimated at 0.8% of B_0 (based on composting emissions of 0.4 kg CH_4 / tonne of green waste [274], and DM content and ultimate methane yield of the organic material as shown in **Table 2-2**).

The mean CH_4 collection efficiency at the baseline landfill is estimated at 75% of total lifetime methane production [275]. The CH_4 collection efficiency is often greater than 90% for a closed landfill cell. However, the collection efficiency from active cells is about 35% [276]. If CH_4 is produced from the easily degradable organics (e.g. food) before a cell is covered, collection efficiency for food and green waste could be low. Thus, for the Landfill Leaky case, mean collection efficiency is assumed to be 50% (distributions shown in **Figure 2-5**).

Nitrous oxide (N_2O) emissions are assumed to be 0.2% of the N in the material from landfills (both) and AD, and 0.5% from composting. Distributions are shown in **Figure 2-6**.

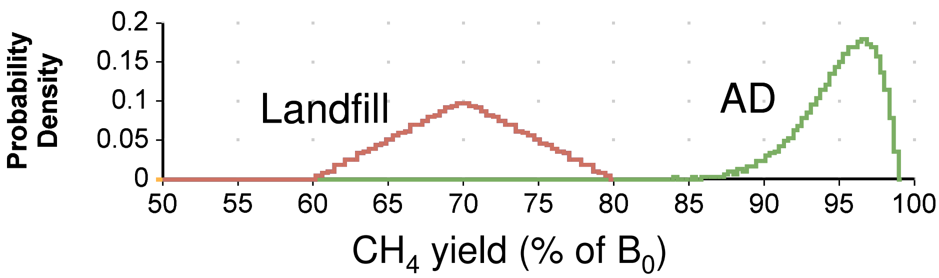

Figure 2-4. Probability density functions assigned to methane (CH_4) yield obtained from organic material placed in a landfill and anaerobic digester (AD); x-axis corresponds to methane production in percent of the ultimate methane yield (B_0). Mean CH_4 production from composting is 0.83% of B_0, with a triangular distribution of 0.2, 0.3, 2.0 for lower limit, mode, and max, respectively (not shown on chart).

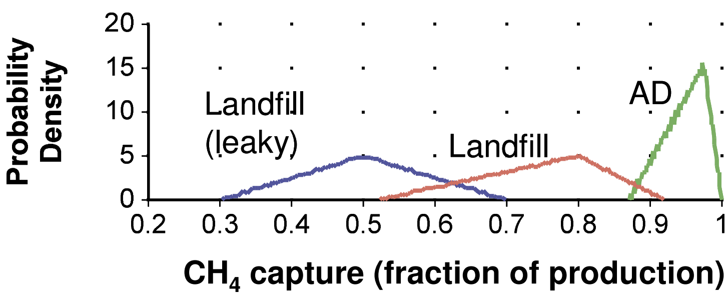

Figure 2-5. Probability distributions assigned for the methane collection efficiency of Leaky landfill (left most), landfill (middle) and AD (right). Mean values are 0.5, 0.75, and 0.95, respectively. For the landfill, collection efficiency includes methane oxidized by cover material. Note that (1- collection efficiency) corresponds with the fraction of CH_4 produced that is emitted to the atmosphere (often called "fugitive emissions" in the context of an AD facility).

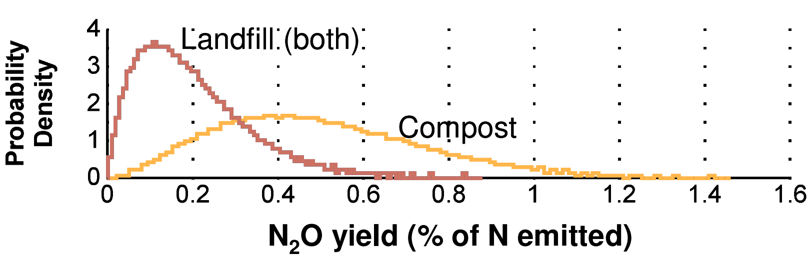

Figure 2-6. Probability density function of the percent (per 100) of N present in the organic byproduct emitted to the atmosphere as N_2O. Averages are 0.2% for the landfills and AD, and 0.5% for compost [272, 277, 278].

2.3.4 Use

Methane-rich gas is produced from the anaerobic microbial conversion of organic matter. This landfill gas (biogas) is usually 45%-65% methane, with the remainder largely CO_2. This gas may also contain hydrogen sulfide (depending on the nature of the organic material), ammonia, and siloxanes[4]. The degree to which the gas must be cleaned before use depends on the sensitivity of equipment to impurities, maintenance frequency, and desired device longevity.

Of the total amount of gas collected at a landfill (see below), 90% is used to generate electricity [279] at 35% efficiency (HHV). The remaining 10% is flared. Of the biogas generated at the AD plant, 5% is lost as fugitive emission; 95% of the collected biogas is used to generate electricity (35% HHV efficiency), while 5% is flared. Electricity required for gas use amounts to 5% of electricity generated from landfill gas, and 15% at an AD facility – the parasitic load is higher for the AD case because of requirements to process the feedstock and effluent [280]. Half of the heat available from the cogeneration equipment at the AD facility is assumed to be utilized, offsetting natural gas use (60 g CO_2eq/ MJ (HHV) delivered [123]). Electricity production is credited with 650 g CO_2eq/ kWh, the average GHG intensity of electricity generation in the US.

Both the composted and digested material can be used for soil amendment and fertilizer. This soil amendment may be used in agricultural, decorative, or structural applications, and is assumed to offset N and P fertilizer production, whereby 50% of the P present in the original byproduct is available for use from compost and AD digestate, 25% of original N of composted material and 50% of original N content of digested material are available for fertilizer.

[4] Siloxanes are silicon-containing compounds and are used in many cosmetic products. Upon being degraded anaerobically in the landfill, they may become entrained in the landfill gas.

2.3.5 Storage

When the organic material is "in-place", some of the C contained in the material will persist in a solid form, and thus not be returned to the atmosphere. If a certain process or practice protects organic C from being oxidized to CO_2 and entering the atmosphere, this action will delay the GWP associated with the C for as long as the storage is maintained. The degree to which organic C is considered sequestered depends not only on time horizon, but also on how the material is being stored and/or put to use. Long-lived products like structural wood (buildings, furniture) may be in use for over 100 years, during which time the bio-C is "protected." The recalcitrant component of compost may resist oxidation for weeks or months [281], while pyrolyzed organics (char) added to soil may stay in-place for centuries or millennia [282].

Of the C contained in the organic material, 28% of the C remains for the Landfill leaky scenario (**Figure 2-7**). These values are derived from the carbon storage factors for food waste, grass, branches and leaves, as reported by Barlaz [283]. Compost and AD residue is assumed to be applied to land, and so no C storage is assumed for either. If organic material is pyrolyzed (to make char) and added to soil, the fraction of C sequestered would be higher.

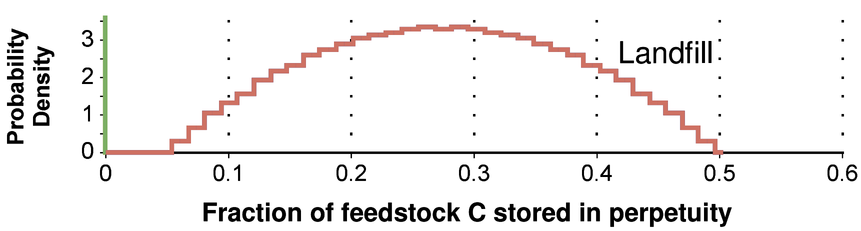

Figure 2-7. Probability density assigned for the fraction of carbon contained in organic byproduct that is sequestered indefinitely in both landfill scenarios. No permanent C storage is assumed for compost or AD.

2.4 Results & discussion

Results shown in **Figure 2-8** indicate that the overall GHG impact of organic byproduct management using AD is slightly lower than the impact of the landfill (baseline case) and compost. However, if the landfill gas collection efficiency is low (as with landfill leaky), the impact of CH_4 from a landfill is high. The degree to which C is sequestered in landfills could also shift the landfill system from a net positive to a net negative GWP impact, as shown for the landfill case in **Figure 2-9**.

In order to gain some more insight, consider the value added from anaerobic digestion, compared to landfilling, as well as the increased management costs. The AD system delivers 220 kWh of electricity, 0.6 GJ of heat, 14 kg of N and 1.5 kg of P, while the landfill system delivers 120 kWh of electricity. Assuming electricity is worth 8 ¢ per kWh, heat is worth $8 per GJ, and the nutrient values shown in **Chapter 4**, the products from AD are worth $15 more per tonne of waste than the electricity revenue from landfilling (baseline). Thus, AD does add value, but it is unlikely that the more involved processing requirements of the AD are within $15 of the per-tonne cost of composting or landfill disposal. If CH_4 emissions from landfills are high (as with the landfill leaky scenario), the GHG benefit of AD amounts to an additional $20 per tonne of waste (with a price of $25 per tonne of CO_2eq). In addition, AD may enable more local processing, thus reducing transport costs.

From these estimates, it is clear that human-produced waste is a minor constituent of the GHG emission in the US (the magnitude of emissions from animal manure is similar, and are analyzed in **Chapter 4**). Nonetheless, managing organic byproducts is an important activity because of both the potential for poorly managed organics to be rank and unhealthy as well as the potential benefits in terms of soil amendments and energy harvest. While GHG credits could supply a relatively small amount of income in these systems, it seems likely that other concerns including desirability and availability of landfills, non-GHG pollutants, and demand for compost will drive the evolution of organic waste utilization in the US.

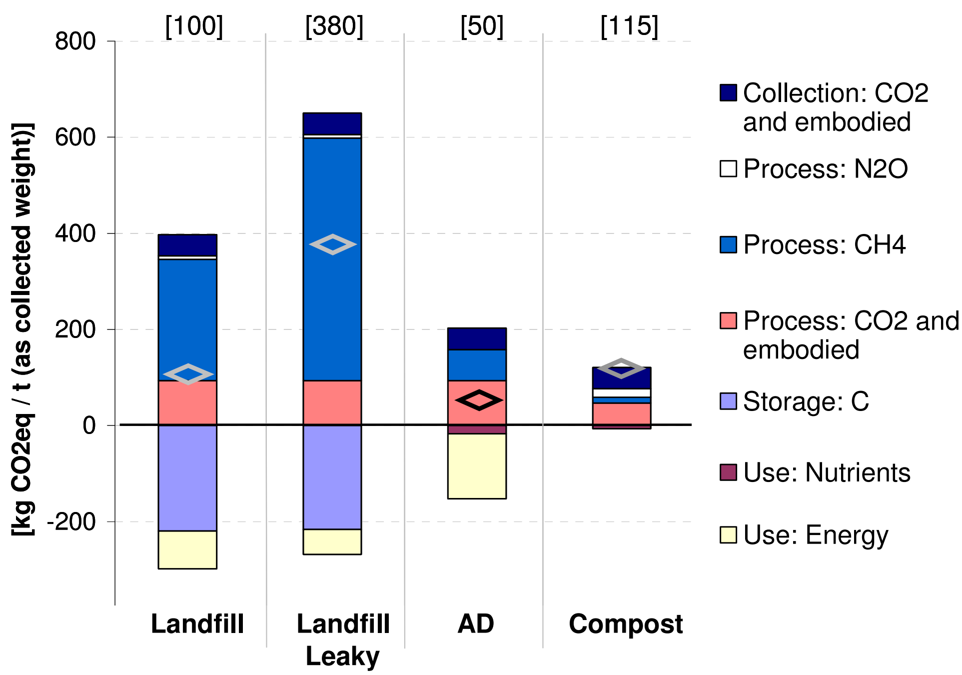

Figure 2-8. Results of the simplified GHG comparison of four organic byproduct management systems (Landfill good, landfill bad, anaerobic digestion (AD), and compost). The values in brackets and the diamond shapes both indicate the net GHG impact of managing one tonne of organic byproduct (as-collected weight).

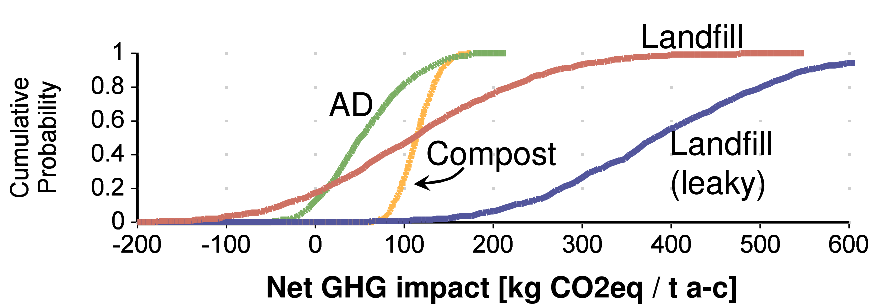

Figure 2-9. Cumulative distribution of the net GHG impact of the four organic byproduct management systems, in unit [kg CO2eq / tonne as-collected weight]. The wide distribution of the net impact of landfills is a result of uncertainty regarding methane collection efficiency and C sequestration.

2.5 Appendix

2.5.1 Discussion of estimates in Table 2-1.

Kim and Dale estimate global potential of crop waste and byproducts to supply energy and find about 80 Tg dry matter of wasted crops and 1500 Tg DM of crop residue potentially available [60]. They use crop production and crop waste estimates from the FAO of the United Nations and assume enough crop residue is left in the field to provide a 60% cover.

Campbell et al. use historical land use data, satellite-derived land cover information and ecosystem models to estimate an area of 385 – 472 M ha of degraded globally with an average above-ground NPP of 4.3 t / ha (and assume 20 kJ / g). [57]. Field et al. use similar methods to estimate 386 M ha of globally abandoned agricultural land (with considerable uncertainty), with average yield of 3.6 t / ha / yr (45% C content), can provide ~27 EJ / yr (20 kJ / g biomass) [8].

The NAS report presents estimates of biomass resources available that do not "compete for land on which an existing crop is produced for food, feed, or fiber or compete for pasture land that will be needed to feed a growing and increasingly affluent population, even with yield increases", produced in such a way so that "the environmental impact on land used for biomass production should be no worse than that of its previous use and provide greater benefits wherever possible" (p70) [218]. 76 Mton of corn stover (of 370 Mton stover produced); 15 Mton of wheat and grass straw; 15 Mton from hay (10% of yr2003-2007 average production); 104 Mton from dedicated crops (24 M acres of CRP land available for dedicated energy crops, out of 35 M acres); 60 Mton dead trees; 41 Mton forest product byproducts (33 used; 8 available); 15 Mton of urban wood (not including 8 Mton already used and 13 Mton MSW); 6 Mton of manure (dry, 10% of 60.6 Mton); 90 Mton of MSW (2/3 of 140 Mton potentially usable). Summary (yr2008) says 110 Mton "woody residues" (60 Mton dead; 40+8 forestry industry byproducts (used and unused); 15 urban wood. Total is 416 Mton in 2008; 548 Mton in 2020.

According to the DOE report by Perlack et al., unexploited current sustainable biomass supply from forests includes: 41 Mton from logging and other removal residues, 60 Mton from "fuel treatments", 8 from wood residues (forest products industry), and 28 from urban wood residue [48]. Current sustainable biomass supply from ag lands includes: 113 Mton crop residues (75 corn stover, 11 wheat straw, 6 other small grain, 21 "other"), 15 Mton grains for biofuels, 35 Mton manure, 31 Mton "other residues" (including processing, MSW, fats and greases). With high yield increases, the report imagines 377 Mton from perennial energy crops, 428 Mton from crop residues (256 Mton corn stover, 52 wheat straw, 25 small grain residues, 48 soybeans, 47 other crop residues), 18 Mton from CRP, 87 Mton grain for biofuels, 44 Mton manures, 44 Mton other residues. For this high-biomass scenario, 60 M ha of perennial crops are added, at the expense of active cropland (25 M ha), CRP (10 M ha), and cropland pasture (25 M ha).

Tilman estimates a potential organic product supply of 270 – 430 Mton is available in the US (80 – 100 Mton of dedicated energy crops on least productive and sensitive land; 100 – 150 Mton from crop (corn and wheat) residues; 70 – 140 Mt from woody byproducts, including forestry slash and thinnings, urban waste wood, and wood mills; 20 – 40 Mton from paper/board waste) [243]. Rough global estimate: 500 M ha degraded land globally, providing 90 GJ / ha [58].

Hoogwijk et al. estimate current and future biomass supply globally by using literature and by modeling future food demand and crop yields. Scenarios with high crop yields offer the possibility of large potential biomass supplies. [55]

2.5.2 Details for yard-sourced organic byproduct

Table 2-3. Definition of the yard-sourced organic residue. The representative mix is composed of grass, woody clippings and leaves. The properties reported here are illustrative, as compiled from a few sources [10, 268, 270]

		MIX of yard residue	Grass	Woody	Leaves
Mix make-up	[% of mix DM]	100%	50%	30%	20%
Dry matter	[% of as-collected mass]	55%	45%	71%	56%
Volatile solids	[% of DM]	86%	82%	87%	93%
Methane yield	[L / kg VS]	149	189	135	78
N	[g N / kg DM]	10.9			
P	[g P / kg DM]	1.5			

2.5.3 Discussion of LCA framework

Feedstock production/acquisition

The feedstock may be a harvested crop (including tree harvesting), a processing co-product, a collected residue, or even a waste. The economic value of the feedstock is dependent on location, timing, and perspective (i.e. where system boundaries are drawn). In some cases, a byproduct may truly be a waste, if it does not offset production in any other systems. In the case of manure, some is used as a soil amendment, so its absence would likely mean increased fertilizer purchases, or decreased crop production (or both).

Impacts associated with biomass feedstock acquisition should account for emissions from:

- Land-use-related depletion of bioC stocks (direct or indirect, see below for discussion)
- Fuel use for production and harvesting
- Production and transport of chemical inputs (e.g. fertilizer and pesticide)
- Production and transport of farm machinery and supplied
- N_2O and CH_4 emissions arising directly from, or as a consequence of, the feedstock production

A residue (byproduct) is typically considered a secondary output from a production process. If an organic residue is truly a waste, it does not carry any economic value, and it does not offset supply of any other good or service [224]. In this case, there are no upstream impacts associated with the biomass residue.

On the other hand, if the residue does have economic value, it will be considered a co-product that supports the production system, and impacts associated with the co-product should be estimated using standardized LCA procedures.

If the system producing the main product and residue are not separable, the system expansion method is a good way to estimate life cycle impacts. With system expansion, all impacts of the product system are assigned to the main product, as are any "credits" associated with co-products. The credits are derived from reduced production of a good with similar qualities as the co-product(s). The animal feed co-product from corn-based ethanol production is a good example, where ethanol is the primary product, and the feed co-product reduces demand (and subsequent production) of other animal feeds. Fundamentally, the system expansion method entails estimating market-mediated changes in production of goods caused by the availability of a co-product(s), and should only be used when assessing small changes in demand.

Impacts can be allocated among co-products using the fraction of total economic value embodied in each co-product. This method takes an attributional (average) approach: the fraction of total economic value from each co-product is assumed to be indicative of the fraction of the total impacts attributable to each co-product. When a co-product has an economic value equal to or less than zero, there should be no impacts of production assigned to its use.

Biomass production/acquisition may be associated with a change in the stock of terrestrial carbon (C). The time scale considered, alternative uses for biomass, and current practices all influence the degree to which biomass utilization is associated with a change in biological carbon (bioC) stock. A permanent change in terrestrial bioC stock is associated with a change in the atmospheric CO_2 concentration (unless the time scale is expanded for millennia, at which point the atmospheric and oceanic C pools are in equilibrium [108]). Thus, the degree to which utilizing biomass does or does not increase atmospheric C depends critically on the baseline assumed for a given bioC pool: if biomass utilization causes a chronic depletion of the C stock compared to the baseline, the products and services provided by that biomass should be assigned land use impacts; likewise, if the biomass production is associated with increased bioC stock, the biomass produced should be given credits. It should be stressed here that the *baseline* bioC stock defined for particular area producing biomass is critically important.

For crops harvested (at least) annually, it is easy to conceptualize the bioC as being part of a short-term C cycle: for a particular plot, new growth “replaces” the bioC harvested previously. Thus, the bioC stock will recover to its pre-harvested magnitude within the next year. Situations where the harvested C is used to supply energy services (and the C is oxidized and released to the atmosphere) are typically considered to be climate-neutral, since the time frame over which the terrestrial stock of bioC is replenished is so short. However, it is important to recognize that LUC in the (perhaps distant) past may have occurred on the land harvested annually, and so the annual cropping could be associated with chronic depletion of the bioC stock.

The bioC stock depletion associated with harvesting an old forest may take centuries to be rebuilt if the forest is allowed to regrow. This example illustrates the importance of long-term management when considering the climate impact of biomass. A plot of land would have to be managed for centuries to avoid chronic depletion. In this case, the bioC stock may be modeled by instead assuming a small annual harvest equal to the growth of the C stock. In practice, old forests are not usually harvested and allowed to grow undisturbed for centuries.

When the bioC stock is reduced (e.g. through deforestation), and the subsequent land use is altered from the baseline land cover (e.g. forest converted to pasture or cropland), products and services which motivated the land use conversion are associated with a transfer of C from a bioC stock to the atmosphere. There is currently no agreed-upon method with which to allocate these emissions. A simple method would be to allocate emissions over a 20-year period following land conversion (e.g. [96]). Similarly, Zaks et al. allocate over 20 years, but use a linear declining allocation such that responsibility is weighted toward the years following allocation, declining linearly to zero in the 21st year [284]. Another possibility is to allocate emissions among all the products and services supplied by the land under consideration since conversion.

Equilibrium economic models are also used to estimate the land-use implications of policies regarding biofuels (see **Chapter 1** for discussion).

3 Life cycle assessment of animal products in the United States

Abstract. Animal farming involves converting fodder into animal products, and animals are the primary consumers of harvested crops globally and in the US. Roughage supplies 75% and 85% of feed in the US dairy and beef industries, respectively; grazed biomass makes up 55% of total feed consumed during beef production. Pigs and poultry are mainly raised on corn and soy-based concentrate feed mixes, with various supplements and feed meals. A life cycle assessment is performed of the global warming potential (GWP), water demand, and land use of animal production in the US. The goal of this LCA is to synthesize various estimates of feed consumption by animal categories and allow an integrated assessment of animal production in the US. I find that gross GWP emissions from animal production (excluding considerations of land conversion) is just over 350 Tg CO_2eq in the US for yr2007, which is about 5% of gross US emissions. The greenhouse gas (GHG) impact of producing beef is more than six times the impact of chicken meat. Methane emissions from cattle outside of feedlots make up 70% of the total climate impact of beef. Nitrous oxide emissions are responsible for 35% of the GHG impact of poultry meat.

3.1 Introduction

Human-managed agriculture exerts a major influence on ecosystems worldwide [285, 286]. About 34% of global land is used for agriculture by humans, of which 65% is grazed by livestock [77, 287, 288]. Animal agriculture is responsible for a large share of human-induced impacts on nutrient cycles, water quality and flows, and land use patterns [289-293]. The purpose of this chapter is to synthesize and compile fodder use data and estimate farming impacts during animal product (AP) production.

Animal management schemes are diverse globally, but a common reality is that livestock are allowed access to biomass (if not fed outright) when local market conditions are such that the biomass is not more valuable as food, fuel, fiber or fertilizer. Bouwman et al. estimate that over the period from 1970 to 1995, the amount of fodder consumed worldwide by livestock (dry mass) grew by about 40%. The authors also project that total fodder consumption will grow by another 40% between 1995 and 2030. Grass accounts for about half of this total [294]. Animal production consumes 60% of human-appropriated biomass flows globally, in terms of dry matter (DM) [30].

The global distribution of AP consumption (as of yr2002) is not uniform (shown in **Figure 3-1**). The one billion people (16% of global population) living in the highest per-capita meat consuming countries accounted for over 40% of total meat consumption, while the least-consuming 3.1 billion (50% of global population in yr2002) consumed just 15% of the total global meat production. The degree of meat-eating is certainly influenced by cultural and/or religious factors. In addition, uniform rates of global meat consumption at the high levels of the US or Europe is not an eventuality, due to the persistence of "social wrinkles", which maintain broad distributions of consumption [295]. However, the non-uniform distribution of meat consumption shown in **Figure 3-1** illustrates the substantial potential for increasing demand worldwide.

Global trade of agricultural products means that consumers are at least partly responsible for environmental impacts and resource utilization associated with production that occurs in different locations [289, 296-298]. Pelletier et al. capture some of the interconnectedness of global agri-products when they explain that "…European salmon production is partially underpinned by Brazilian soy production, while Chilean salmon consume European poultry processing coproducts [which European salmon producers are banned from using] that were themselves produced using Brazilian soy." (p 8734) [299].

Smil reviews meat consumption by humans and the large-scale issues associated with animal agriculture, namely greenhouse gas (GHG) emissions, manure pollution, impacts of crop production and poorly managed grazing lands, and the prospect for increasing meat consumption worldwide [300]. Smil also estimates that the global zoomass of domesticated vertebrates is ten times larger than wild vertebrates [3].

In addition to more efficient fertilization and irrigation for fodder production, integrating plant-based proteins with processed meats (e.g. in ground beef and sausage) could reduce meat demand and lower the aggregate detrimental impacts of meat production [300]. Similarly, many researchers have shown that GHG emissions from poultry and dairy products are lower than from beef (see **§3.7**), and so replacing higher-impact AP with less demanding AP (e.g. replace beef with chicken), can lower impact as well.

Reijnders and Soret perform an LCA of some meat and plant-based food and report that replacing meat protein with soy protein can halve the environmental impact of a meal. Meat demands about ten times the land and water resources as plant-based protein [92]. However, water consumption for some meat alternatives (e.g. nuts) may also be quite high.

In the "Livestock's Long Shadow" report from the United Nations, Steinfeld et al. estimated that carbon dioxide (CO_2) emitted via land use change (LUC) is the largest constituent of the global warming potential (GWP) of livestock products and services globally (**Figure 3-2**). This emission of biospheric carbon attributable to livestock (2.5 Gt CO_2/ yr) was responsible for about 40% of LUC from all sources. The total GWP impact of global livestock is 7 Gt CO_2 equivalent (CO_2eq, 100 year timeframe) [301] and amounts to roughly 17% of total, annual anthropogenic GWP (estimated at 41 Gt CO_2eq/ yr in yr2000; see **Chapter 1 Appendix, §1.10** for GHG emission data). The fraction of total GWP from animal production globally will likely decrease if fossil fuel use continues to increase, as it did during industrialization of the US [302].

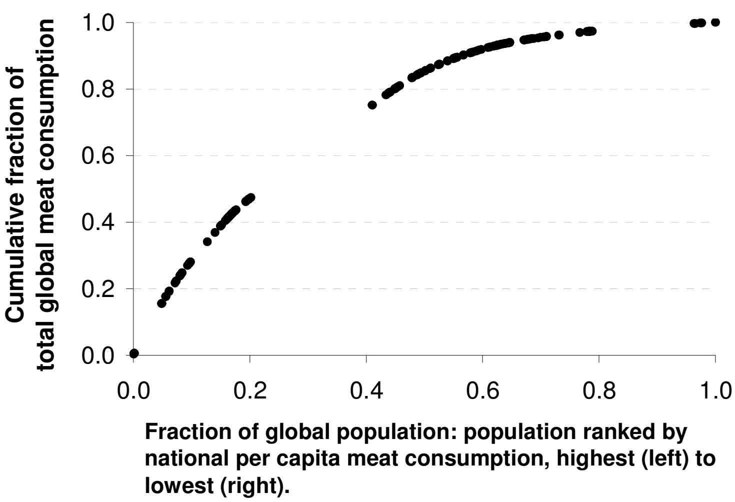

Figure 3-1. Cumulative distribution of national-level, population-weighted global meat consumption in yr2002. Estimated by the United Nations [287], via the World Resource Institute [303]. The figure ranks countries by per capita meat consumption, with highest consuming countries on the left. The y-axis shows the fraction of total global meat consumption, as you include all the countries of the world (in order of decreasing per capita national meat consumption). For example, the 20% of the global population living in the high per-capita consuming countries account for over 40% of total meat consumption.

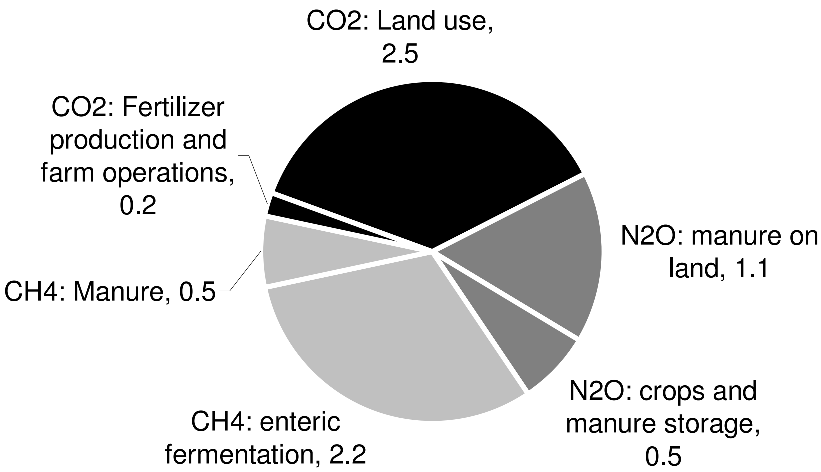

Figure 3-2. Global GHG emissions due to livestock, around yr2000 [301]; units in [Gt CO_2eq/ yr]. The total is about 7 Gt CO_2eq/ yr. For reference, CO_2 emissions from fossil fuel combustion and cement making were ~25 Gt CO_2/ yr [16, 103, 222].

Animal production schemes can be managed to improve the environmental performance of animal farming. Intensively-managed integrated crop-livestock systems offer promise for improved economic and ecological performance [304], but labor-intensive management requirements of such systems have led to the more specialized production in the US [305]. Improving the quality of pasture (through infrastructure like irrigation and fertilization and practices like well-managed intensive grazing) can reduce the climate impact of meat production by (*i*) increasing terrestrial C stocks [78], and (*ii*) by improving the yield and increasing the digestibility of grazed biomass consumed, thus increasing feed conversion efficiency and reducing the fraction of the feed converted to methane [306]. Improving grazing and crop management practices offers the largest GHG mitigation potential in agriculture [78].

Garnett argues that LCA of animal based products should include indirect land use impacts, and consider opportunity cost of various land uses. In addition, the degree to which humans need to consume AP at all is offered as another important perspective with which to view impacts of livestock production and mitigation options [307].

Improving pasture biomass yield and quality can reduce impacts of cattle. Similarly, increasing intensification can reduce the land required for producing cattle, since high-yield (perhaps genetically modified) crops can be fed for high-efficiency feed conversion. If pasture-based meat yields are lower per area than confined feeding systems (as modeled in [308] and [309]), then switching to pasture-based meat without reduced meat consumption rates could induce indirect land use change (see **Chapter 1, §1.8** for discussion of land use).

Johnson and Johnson [310] review the GHG inventory from animal agriculture in the US. They suggest including fuel and nitrous oxide (N_2O) emissions associated with cropping in the agricultural category. This inclusion would increase total agricultural emissions from 486 to 670 Tg CO_2eq/ yr in yr2005, and the contribution from livestock would be 380 Tg CO_2eq/ yr.

Since animal agriculture is currently a large influence on the biosphere, and since there is apparently large potential for AP demand to increase, the continued expansion of animal agriculture is a topic that merits attention, and the environmental impact of feeding these animals should be understood. The goal of the current chapter is to provide a foundation on which to assess the potential for organic energy harvest to offset impacts of animal production (see **Chapter 5**), as well as to identify important considerations regarding biomass appropriation for AP and related land use.

In this chapter, I estimate the biomass use, land and water use, and GHG impact of animal production. I first review the magnitude of AP consumption in the US (**§3.2**) and trace gas measurement (**§3.3**) and methods of agricultural life cycle assessments (**§3.4**). I then synthesize and compile fodder use data and calculate the impact of crop production and animal operations (**§3.5**) in order to determine GHG impact of current AP systems in the US(**§3.6**).

3.2 Animal product supply and fodder use in the US

Livestock in the US eat a mixture of grains, grazed biomass, harvested roughage, grain-plant silage, and other concentrate feeds. Roughages, including grazed biomass, make up the majority of animal feed on a dry-mass basis. About 340 Mt DM of roughages and 170 Mt of DM of concentrate feeds are consumed by livestock in the US annually (**Figure 3-3**). Concentrates include grains (e.g. corn, wheat and barley grains), meals (e.g. soymeal, alfalfa and animal meal) and milling byproducts (e.g. wheat mill feed, corn gluten feed, and ethanol distillers' grains).

Corn makes up 65-70% of concentrate (non-roughage) feed in the US (**Figure 3-4**). In yr1990, animal feeding used 58% of total corn grain production, compared with 45% in 2008. Other grains used as fodder include wheat, sorghum, barley and oats; these four grains made up 17% in yr1990, compared to 6% of concentrate feed in yr2008. Co-products of biomass processing make up over 20% of concentrate feeds (DM basis), and include: soybean meal (co-product with soybean oil); corn, wheat and rice processing co-products; animal protein feeds (animal bits from slaughter and processing facilities); and sunflower, cotton and other seed meals. Exports of whole soybeans from the US have recently been at 40% of production (yr2007 & '08); when soymeal and oil are included, over 50% of the US soybean DM production was exported in those years.

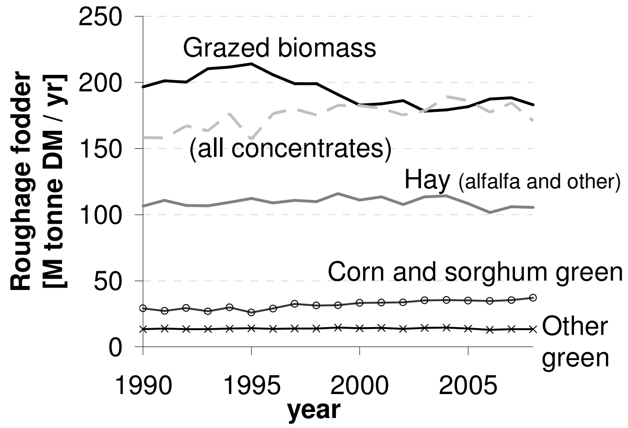

Figure 3-3. Fodder DM consumed by livestock in US. Grazed fodder is from [43], converted from corn equivalents to grazed dry matter using a conversion factor of 1.50 kg DM grazed roughage/ kg DM corn equivalent (adapted from National Research Council [46]). See §3.5.3 for details and data and Figure 3-4 for "all concentrates".

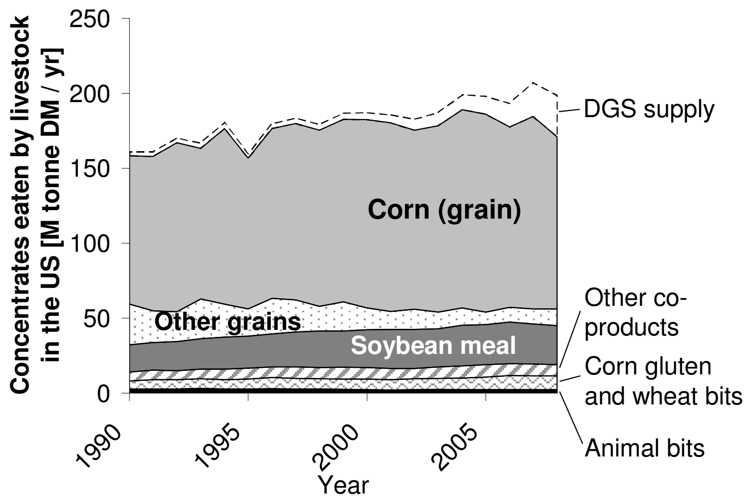

Figure 3-4. Concentrate livestock feed in US, dry mass (DM) basis. As-fed mass from [214], converted to DM using factors shown in Appendix (Table 3-25). The "DGS supply" category shows an estimate of the ethanol co-product feed supply, calculated by multiplying the mass of corn (DM) used to produce fuel ethanol [214] by 0.35 (DGS DM produced for every mass unit of corn grain DM entering an EtOH plant [311]). "Other grains" includes wheat, sorghum (grain), oats and barley; "Other co-products" includes alfalfa, canola, cottonseed, rice and sunflower meals (among others); "Corn gluten and wheat bits" includes corn wet-milling co-product feed and wheat milling co-product feed. See Appendix §3.10 for details.

3.2.1 Cattle diet and food borne illness

Cattle diet and fodder processing may influence the microbial community in the digestive system and the prevalence of food borne illness. Feeding distillers grains has been associated with increased incidence of virulent *E. coli* O157:H7 in cattle [312-314]. Likewise, steam-flaked corn has been also associated with increased rates of O157:H7 shedding, compared to rolled corn [315]. Other studies show that inclusion of distillers grains has no impact when the fodder mix is based on steam-flaked corn [316, 317] (although [316] reports relatively low *E. coli* shedding rates overall). The diversity of study results indicates that diet alone cannot explain prevalence of O157:H7 [318, 319]. One mechanistic theme, however, involves reduced starch levels entering the hindgut compared to when whole corn is fed. Starch in the hindgut leads to acidic conditions, which tend to destroy the *E. coli*. When feed with less starch is fed (as with distillers grains), or if starch is more completely digested before reaching the hindgut (as with steam-flaked corn), the hindgut is less acidic, perhaps allowing increased *E. coli* O157:H7 survival rates through the digestive system.

3.3 Measuring and modeling CH_4 and N_2O emissions in agriculture

Methane (CH_4) and nitrous oxide (N_2O) accounted for roughly 17% and 7%, respectively, of the warming potential of the global GHG emissions in yr2005 (based on 100yr GWP; see **Chapter 1 Appendix, §1.10**). In the US, CH_4 and N_2O together make up 12-13% of the GWP of all national GHG emissions. Methane from livestock and nitrous oxide from soils represent about 2% and 3%, respectively, of the GWP associated with all anthropogenic GHG emissions in the US in yr2007 [102].

3.3.1 Methane

Human influence over the CH_4 cycle may have begun over 10k years ago through ruminant megafauna predation, with subsequent CH_4 emissions reductions causing cooling [320]. Pre-industrial CH_4 emissions were likely between 200 and 250 Tg (CH_4) per yr, with the anthropogenic fraction at 10-20% of the annual total. Currently, natural wetlands are still the largest single CH_4 source, but anthropogenic sources make up about 60% of all CH_4 emissions. Energy-related CH_4 emissions (mining, distribution and burning) are likely around 100 Tg/ yr, and emissions from ruminant livestock amount to roughly 85 Tg/ yr [103]. Growth of atmospheric methane concentration slowed in the 1990's (perhaps due to reduced natural gas production-related emissions), and the concentration was relatively constant in the yr1997-2006, when anthropogenic emissions were increasing, but emissions from natural wetlands were relatively low [321]. The atmospheric concentration of CH_4 may have been on the rise in yr2007 [322].

As will be shown in **§3.6**, CH_4 is a major part of the life cycle impact of cattle products. Methane emitted by cattle can be measured by placing animals in a controlled atmosphere chamber so gas concentrations can be measured. Another method uses sulfur hexafluoride (SF_6) methane monitoring (gas is sampled from near the animal's mouth, and a bolus placed in the rumen emitting SF_6 at a known rate allows determination of air dilution of the sample, and thus CH_4 emission [323]). Micrometeorological methods (whereby open-air CH_4 concentrations are measured up- and down-wind of the site being sampled) are also used [324]. The SF_6 tracer method agrees well with gas chamber results [325-327].

The type of fodder consumed by ruminants often influences CH_4 emissions. Harper et al. use a micrometeorological method and report that CH_4 emissions are about 8% of gross energy intake for cattle grazing low-quality pastures, and about 2% for animals eating a high-grain diet (80% oats and 20% alfalfa) [328]. On the other hand, Beauchemin and McGinn report that CH_4 emission from beef cattle were similar for high forage diets (barley-silage based) and high concentrate diets (corn-grain based), and were around 6.2% of gross energy intake [329, 330] (see also [331]). Inclusion of supplements like coconut oil can reduce CH_4 emissions from 8% to 6% of gross energy intake [332, 333]. Chaves et al. find that cattle in Canada grazing on either alfalfa or grass had similar emission of CH_4: about 6% of gross energy intake [334]. The largest uncertainty in determining methane conversion factors from grazing is often the estimate of the total feed intake, since grazed fodder cannot be weighed before consumption [327, 334].

To estimate the amount of CH_4 produced by cattle in the US, the EPA uses an IPCC Tier 2 method [244]. Cattle are assigned to a "type" category and a "region" category. Two parameters are defined for each subgroup: the digestible energy (DE) of the feed consumed by animals in the category, and the methane conversion factor (Y_m), which indicates the fraction of feed energy intake (gross energy) that is emitted as CH_4 during digestion. The amount of feed consumed daily by the "representative animal" of each subcategory is calculated using IPCC equations, which are based on animal type, activity and performance level, and regional climate [157].

The US EPA estimates that ruminant livestock emit 24% of CH_4 emissions in the US. Cattle (dairy and beef) are estimated to produce 95% of these, while horses, sheep, swine and goats emit the rest [244].

3.3.2 Nitrogen and nitrous oxide (N_2O)

Nitrous oxide (N_2O) is a potent GHG when compared to CO_2 on a mass basis. N_2O is an intermediate product in the microbial processes of both denitrification and nitrification [335]. Nitrification is an energy-harvesting oxidation of ammonia and nitrite into nitrate. Denitrification is a reduction of nitrate (and nitrite) where nitrogen oxides present in soil are used as electron acceptors by microbial processes (the electron donor is often organic matter) [336, 337]. Nitrifying and denitrifying microbes are often found together in soil, with denitrification occurring when oxygen (in the *micro* environment) is limited [338]. Distribution of reactive nitrogen and subsequent transformations and ecological impacts are not fully understood [286, 339].

Since dinitrogen (N_2) in the atmosphere is a very stable, non-reactive molecule, many ecosystems are limited (in primary productivity terms) by the amount of reactive nitrogen primary producers are able to obtain. Humans have enhanced reactive N inputs via fertilizer to increase growth of desired plants. As a byproduct, these activities also lead to more reactive N in the biosphere, and thus enhanced emissions of N_2O to the atmosphere.

Contemporary intensive cropping often involves maintaining fields at a state of reactive N saturation. These practices result in “leaky” agro-ecosystems that are consistent N emitters to surrounding environments [340]. N_2O emission rates (measured in g N_2O emitted / g N added) are much higher when N is applied in excess, compared to N application to match plant needs [341]. The N “leaked” from agriculture can enhance N_2O emissions in ecosystems away from the field as well [147]. Nitrogen use efficiency is often less than 50% for cereals [342, 343], meaning that N is typically applied in excess of plant needs. N-use Efficiencies of 65% have been shown using best management practices [344].

Tailoring N application to specific sub-field scale crop and soil requirements through “precision agriculture” could improve nitrogen use efficiency, thus reducing excess inputs of N [285]. These practices can be difficult to implement at the field scale, however, because of gaps in our mechanistic understanding of soil and plant processes, and inadequate information [345].

Remote sensing techniques to assess plant needs show promise to improve N-use efficiency, however, N-use efficiency was <25% when crop yield was maximized [346].

In a review of N_2O emissions from land under agriculture and natural vegetation, Stehfest and Bouwman report that grasslands have lower per ha N_2O emissions than croplands. However, the N inputs for grassland are considerably lower than for crops (and biomass DM yields were not reported) [129]. N_2O emissions from grasslands in the arid west have been measured at between 0.1 and 0.5 kg N_2O -N/ ha/ yr [347-349]. Dusenbury et al. find low N_2O emissions from Northern Great Plains soil under a wheat and alfalfa rotation [348]. Using a micrometeorological technique, Phillips et al. report that N_2O emissions are low (at background levels) when water filled pore space of soil was below 65% or above 95% [350]. Parkin and Kaspar find N_2O emissions from corn and soybeans in the Midwest are three times higher than IPCC emission factors would indicate and are not affected by tillage type [351] while Wagner-Riddle et al. report N_2O emissions are significantly reduced using conservation tillage [124].

In addition to the variability of N_2O emissions indicated by of the range of results reviewed above, measurement techniques can introduce additional uncertainty [352, 353]. Rochette and Eriksen-Hamel conclude that many published studies of N_2O fluxes have <50% confidence in the absolute values reported [354].

Animal manure is the proximate N-source for about half of global anthropogenic N_2O emissions, albeit with a large uncertainty [301, 355, 356]. Davidson [146] finds that estimating N_2O production at 2.5% of fertilizer N application and 2% of manure N excretion matches the growth of atmospheric N_2O concentration between yr1860 and yr2005 better than the method used by Crutzen et al., who estimate N_2O emissions at 3-5% of newly fixed N [147].

Manure management practices and equipment, along with animal management, play an important role in the variability of N_2O emissions. For example, Livesley et al. found that N_2O emission factors were higher in animal "camps" where supplemental feed is offered to animals on pasture [357]. N_2O emission related to applied manure depend on compaction and soil moisture content [358], as well as the nature of the manure itself [359].

3.4 LCA methodology

Life cycle assessment (LCA) is a technique to assess environmental impacts of a product, process or service that includes the impacts from all stages of a product's life span [360, 361]. LCAs are studies of systems and flows; thus, the definition of system boundaries and interactions between flows are at the heart of an LCA. In addition, the practitioner must define the temporal and physio-economic perspective - *i.e.* are we talking about the aggregate, average impact associated with a product system already in existence, or are we trying to determine the impact of a decision or action that has yet to be made? LCAs that seek to determine an average, aggregate impact for a product are called "attributional" [362]; those that estimate the change associated with a decision to produce or consume are called "consequential" LCAs.

Assessments of agricultural systems can be materially affected by the particular LCA methods applied (e.g. [363]). Methods to calculate impacts of co-products in production systems are as follows (in order of preference as defined by the ISO): subdivide the system into subsystems and/or expand the system boundary to include other systems with similar functions as co-products; allocation based on physical (causal) relationships; allocation based on other criteria, e.g. economic [364]. The choice of method depends on the context of LCA, as well as data availability. Farms are complex ecosystems, and our understanding of the interactions and feedbacks between management practices in biomass appropriating systems is incomplete [365].

A consequential LCA (also known as a prospective LCA) estimates the impact of a change in the demand for the functional unit (i.e. the product(s) of interest). This method is appropriate for studying small changes in demand, where the production systems affected by the change in demand are called "marginal systems" or "marginal technologies". As noted by Tillman et al, "LCAs are made because we are concerned about the present and future environmental impact of present-day production and consumption of products. Hence, they [LCAs] should be focused more on the present and future environmental impact, than on the impact that has already occurred." ([366], p 22). The consequential method involves estimating (predicting) the environmental impacts of a decision – and then the estimate can be used to make the actual decision. In this way, consequential LCA is related to economic evaluation using opportunity cost, in which a decision-maker weighs estimates of gained and foregone utility associated with a choice [367]. Note that marginal technologies and processes are not constant through time, and will change as market conditions change (e.g. [368]).

In practice, the attributional method is often backward-looking (that is, it reports the impact of a product that already exists), while consequential studies are usually forward looking, since they deal with the consequences of a decision or action. These temporal perspectives are reasonable, given the motivation for LCA, but they are not fundamental to either method. For example, it is reasonable to ask the question “What will be the average GHG impact of 1 kWh of electricity generated in the US in 2020?” (forward-looking attributional). Likewise, one could perform a retrospective consequential analysis by answering the question “What was the GHG impact of the marginal vegetable oil on the global market in each of the past 100 months?” (backward-looking consequential).

Economic input-output (IO) tables enable the accounting of retrospective economy-wide economic interactions associated with products and processes. Input-output based LCA (IO-LCA) is thus a powerful way to capture representative average impacts and to assess important inputs to production (at the resolution of available economic and environmental data). IO-LCA may also be used to inform prospective LCAs. These IO data represent the average unit of production for a given situation [361].

Markets are international, and accounting for marginal technologies and practices on a global scale can affect calculated impacts markedly, as described by Searchinger et al. for biofuel production [203]. The coverage of IO-LCA has been expanded to account for international flows of goods help to quantify the “virtual” resource consumption and environmental effects associated with meat production and trade [369].

Agricultural production often involves co-products, where a particular system provides more than one product or service. Soybean production is an example of a multiple-function system. Over 90% of the US soybean supply (defined as production + imports – exports – stock increase) is “pressed” [216], yielding the co-products soy oil (often used for human consumption) and soybean meal (often used for animal feed). Other examples of multi-product agricultural product systems include: grain and stalks (e.g. wheat and straw or corn and stover); grain fractions (e.g. starch, oil, gluten, distillers' co-product feed); dairy and beef (whereby some calves and "retired" milking cows are slaughtered for meat); and beef, leather, and meat and bone meal (co-products of cattle slaughter).

An LCA practitioner may want to calculate the impact of one product of a multi-product system (i.e. a production system with co-products). Using the system expansion method within the consequential LCA framework offers a reasonable way of determining the impact associated with co-products. With the change in consumption (production) of the functional unit taken as given, production levels of associated co-products also change. In order to estimate impacts associated with co-products, the system under study is expanded to include systems affected by a change in supply of co-products (systems which perform the same service as one or more of the co-products, and whose production levels are affected by a "decision" to consume the primary product of interest). These affected co-product markets may be called "background systems".

With the system expansion method, the product of interest (defined by the functional unit) is assigned all (gross) impacts of production, but is also assigned the impact associated with co-products. These impacts are often "credits" associated with an increased production of co-products which displace some other production. Consequential LCAs are useful for estimating impacts of small changes in demand [224], particularly since collecting data to specify a marginal technology can be more feasible than calculating an accurate average impact [370, 371].

As an example, consider the consequential LCA performed by Dalgaard et al., using system expansion, of soybean meal feed [372]. Soymeal is produced along with soy oil, and so the decision to purchase soybean meal (in this case, in Rotterdam Harbor) is associated with some production of oil as well. This soy oil, which is not part of the functional unit, nonetheless affects the edible oil market (a background system). The increase in supply of oil due to the decision to consume soymeal thus impacts the production quantity of some other oil; in this case palm oil is deemed the marginal edible oil on the world market. The impacts from soybean production are fully ascribed to the soymeal, but the meal is also given credit for the avoided production of palm oil, since some palm oil production is displaced by the soy oil co-product. Thus, the net impact of soymeal is the impact of the soybean production minus the impact of the marginal palm oil that would be produced if not for the decision to consume the soybean meal. System expansion is also used by Cederberg and Stadig [373] to assess impacts of milk production in Sweden, and by Kim and Dale [374] to allocate impacts of ethanol co-products.

Consequential LCA is useful for estimating changes at the margin, but the very nature of a consequential (prospective) LCA is to predict the future, so choices related to large changes in demand may not be good candidates for consequential LCA. The decision regarding biofuel production mandates could be an example, since the demand profile for corn grain in the US has changed rapidly (**Figure 3-5**).

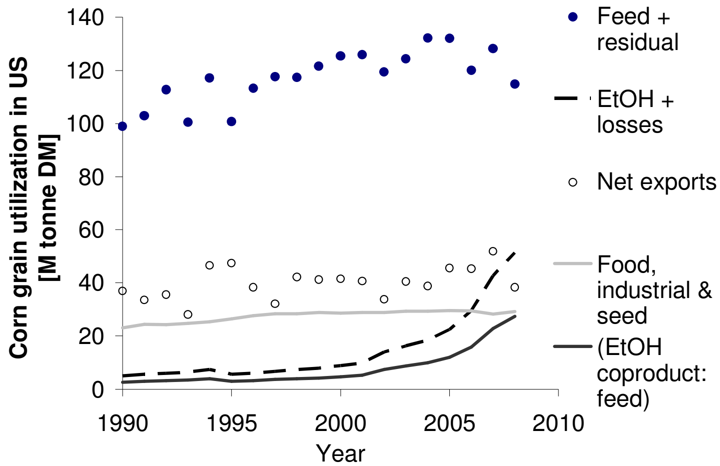

Figure 3-5. Corn grain utilization in the US [214]; "EtOH" = ethanol. "EtOH coproduct feed" is calculated by multiplying corn used for fuel ethanol by 0.35 (DM basis). "EtOH + losses" is calculated by difference. Note that 0.38 kg EtOH are produced per kg corn DM entering a typical dry-mill plant (from [311], and assuming denatured EtOH contains 3% denaturant by volume). Note also that "Food, industrial & seed" does not include corn used to produce fuel ethanol.

3.5 *Methods used here*

This section describes the approach used to estimate land, water, and GWP impacts associated with producing animal products (AP). The system boundary includes feed, fuel and equipment inputs to animal farming, as well as material, fuel, land and water inputs to crop farming and milling requirements during feed production. Impacts from slaughter are not included. For animal fodder ingredients and co-products from the animal farm, economic allocation was used where necessary.

Feeds are quantified here in terms of dry matter (DM) mass. Units of DM mass clearly do not capture the distinctions between animal feeds. As an example, consider the difference between one kg DM of field-dried hay and a one-kg DM mix of corn and soybean meal. Similarly, one kg of corn is not the same to a chicken as an engineered mix of corn, soybean meal, and other feed ingredients. Each feed has particular digestibility characteristics for particular species [46]. The amount of any particular feed ration needed by an animal depends on the species, activity level (including “work” that must be expended to procure food and water), environment (including climate), and genetics. The reader should keep in mind the limitations of using the DM unit, particularly with regard to differences in feeding efficacy. However, the DM unit is used here to offer insight about interactions between animal feeding, energy harvesting, and land use. Fodder flows and impacts are shown for individual feed categories, so the reader can apply relevant feed comparisons as desired.

With recent increases in fuel ethanol (EtOH) produced from corn grain, the supply of EtOH distillers’ grains and solubles (DGS) has increased as well (corn used as an input to EtOH manufacture increased from 4% of corn grain production in yr1990 to over 30% in yr2008 [214], see **Figure 3-5**). DGS include the parts of the corn kernel that are not fermented into EtOH (nor emitted as CO_2 during fermentation). Feeding DGS to cattle and hogs is common in the Midwest [375], but the USDA does not currently track the amount used in feed, so quantities are not well constrained. The “DGS supply” category in **Figure 3-4** represents the amount of DGS likely produced in the US, but does not specify how much is fed domestically. For the rest of this chapter, use of DGS is not accounted for.

3.5.1 Animal products considered

The products considered in this work are milk, chicken meat, beef (cattle and veal meat), pig meat, chicken eggs, and turkey meat. The production (base) unit for meat production is mass of live weight (LW) produced annually; the base unit for milk is mass; the base unit for eggs is one egg. The edible mass, food energy, and protein contents of each product are shown in **Table 3-1**.

Annual production of AP in the US is shown in **Table 3-2**. In the US, per capita supply of protein from poultry, dairy, beef and pigs has increased from around 55 g/ p/ day in the early 1950's to around 65 g/ p daily in the decade after yr2000. Increasing chicken meat consumption has driven much of this trend (**Figure 3-6**).

Table 3-1. Animal products considered in this work (LW = "live weight"), along with edible mass, food energy and protein content [376].

	Dairy	Broilers (chicken meat)	Beef	Pig (meat)	Eggs	Turkey (meat)
(base unit)	[g milk]	[g LW]	[g LW]	[g LW]	[egg]	[g LW]
[g edible / (unit)]	1	0.50	0.41	0.54	47	0.60
[kcal / g edible]	0.64	2.15	2.85	2.63	1.43	1.96
[g protein / g edible]	0.03	0.20	0.17	0.14	0.13	0.18

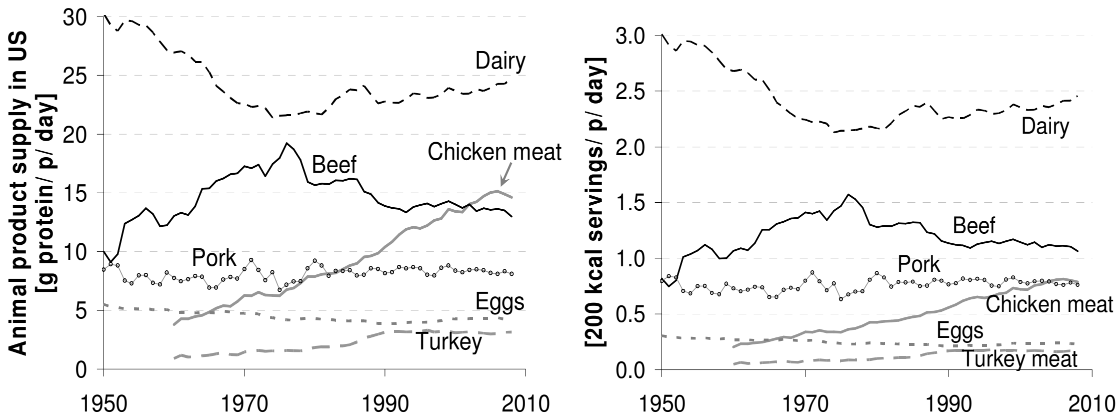

Figure 3-6. Animal product protein (left) and food energy (right) supply in US. Data from [43, 376-379], and does not include net exports or stock changes. Population data from US Census Bureau [47]. See appendix (§3.9) for details. Total protein consumption was 57 and 68 g/person/day in years 1960 and 2008, respectively; total food calorie supply increased from 5.0 servings (200 kcal) to 5.5 servings per person per day.

Table 3-2. Domestic animal production in the US. Milk, broilers (chickens grown specifically for slaughter), beef, pig, table eggs (for human consumption) and turkey production are from NASS (beef and pig production are corrected for live animal trade) [43]; eggs used in hatching are from [380].

	Dairy	Chickens grown for slaughter (broilers)			Beef	Pig	Eggs for eating (table)		Turkey
	[Mt milk / yr]	[Mt LW broilers / yr]	[G hatch eggs / yr]	[Mt LW 'spent' breeders / yr]	[Mt LW / yr]	[Mt LW / yr]	[G table eggs / yr]	[G hatch eggs/ yr]	[Mt LW/ yr]
1990	67	11.6	9.3	0.1	16	9.7	56	0.6	2.7
1991	67	12.3	9.5	0.1	17	10.3	57	0.6	2.8
1992	68	13.1	9.7	0.1	17	10.9	58	0.6	2.9
1993	68	13.9	9.6	0.1	17	10.8	64	0.6	2.9
1994	70	14.8	10.3	0.1	17	11.1	65	0.6	3.0
1995	70	15.5	10.7	0.1	18	11.1	66	0.7	3.1
1996	70	16.5	10.9	0.0	17	10.5	68	0.7	3.3
1997	71	17.0	11.4	0.0	17	10.9	68	0.7	3.3
1998	71	17.5	11.7	0.0	17	11.7	70	0.7	3.2
1999	74	18.5	12.2	0.0	18	11.7	72	0.8	3.1
2000	76	18.9	12.2	0.0	18	11.7	74	0.8	3.2
2001	75	19.3	12.2	0.1	18	11.7	75	0.8	3.3
2002	77	20.0	12.2	0.0	18	11.9	76	0.8	3.4
2003	77	19.9	12.1	0.1	18	11.9	76	0.7	3.3
2004	77	20.8	12.1	0.1	18	12.1	78	0.8	3.2
2005	80	21.7	12.5	0.0	17	12.4	80	0.8	3.2
2006	82	22.1	12.4	0.0	18	12.8	81	0.8	3.3
2007	84	22.3	12.7	0.1	17	13.4	80	0.8	3.4
2008	86	22.6	12.5	0.1	17	14.2	79	0.8	3.6

3.5.2 Impacts related to animal-based co-products

Animal production systems often produce more than one type of output. For the purpose of this LCA, I have classified animal co-products into types, which I describe as “reproductive” and “processing”. The former arises from the animals’ reproductive cycles. For example, breeding chickens produce eggs, which are hatched and grown to slaughter (these birds grown specifically for slaughter often called “broilers”). The breeding stock may be slaughtered for meat, and thus is a co-product with broiler production; likewise with pigs and turkeys. Dairy cattle produce some “extra” calves (not needed to replace the milk-producing cows) as co-products of milk, and the dairy cow itself becomes a co-product of milk production when it is “retired” and slaughtered. Similarly, hens producing eggs for human consumption are often slaughtered for meat products. The other co-product type (“processing”) is associated with animal processing after slaughter. In addition to the primary meat product of slaughter, co-products include “specialty meats” (e.g. tongue, stomach, feet) and edible fats, along with hides (for leather), and other animal bits, often used as fodder (feather meal, meat and bone meal and fats) (**Table 3-3**).

For beef, pork, and chicken meats, the reproductive co-products are aggregated with the animals grown primarily for meat. Thus, the production of “one kg LW of beef” includes steers and heifers grown specifically for slaughter, as well as “retired” cows and bulls which produce the calves for growing. Likewise, “chicken meat” will include young chickens as well as older breeder chickens. The ratio of co-product meat to the primary product depends primarily on the reproductive lifespan of breeding stock, and is shown in **Table 3-3**. The co-product meat is typically less valuable than the primary meat product, and may be used in factory-processed meats or pet food. Likewise, some parts of the primary product (meat from an animal grown specifically for slaughter) are worth more than others – for example, light breast meat may command a higher price than darker leg meat from the same chicken. However, for the sake of simplicity and tractability, these live products are aggregated to a common unit of live weight slaughter, which can be converted to an edible fraction and food energy (or protein) supply.

For dairy and eggs, the reproduction-related meat co-products cannot be so readily aggregated with the primary products (milk and eggs). In these cases, system expansion is used to credit the primary AP with some offset meat production (see **§3.4** for description of system expansion). For the extra dairy calves, the marginal meat is assumed to be beef, since these animals may be raised for veal, a relatively high-value product. The marginal meat displaced by retired cows and laying hens is assumed to be chicken meat (kcal edible basis) since dairy cows provide lower-value meat than purpose-grown beef. Furthermore, chicken meat production has been increasing, and so is considered the marginal meat, which would supply the meat calories if not for the slaughtered milk cow.

Production of co-products from dairy and egg production is shown in **Table 3-22** (**Appendix**, **§3.9**). The number of dairy cows slaughtered is reported in [43]. Live weight (LW) at slaughter is calculated assuming the slaughter weight of a dairy cow is the same as a beef heifer. The number of extra calves from dairy production is calculated by subtracting the calves necessary for replacement (see **Appendix**, **§3.9**) from the total number of dairy calves produced. The number of dairy calves produced is calculated assuming a calving cycle of 13.2 months, a 6.5% stillbirth rate, and a 7.8% mortality rate for live-birthed calves [381]. The extra calves associated with dairy production are credited with 150 lb of beef LW production. The total number of 'old chickens' ('spent' reproducing chickens) slaughtered, along with per chicken LW, are given by NASS [43]. About 84% of egg-laying chickens are in the table-egg production system (the rest are in the broiler production system) [380], and this percentage is assumed to represent the fraction of total 'old chicken' slaughter that originates from the table-egg producing system.

For slaughter-related co-products, impacts are allocated to either meat or "byproduct" based on economic value. The carcass, retail and boneless (edible) fraction of animals slaughtered is from the USDA [43, 377, 378]. Byproduct value is shown in **Figure 3-7** for beef and pork, as a fraction of the total wholesale post-processing animal value. Similar data is not published for poultry, perhaps because the poultry industry is highly vertically integrated, and so all of the value added associated with animal production, slaughter, and processing to retail products often occurs within one organization. Because of this integration, there is a high degree of recycling and value capture in the chicken processing industry. While the byproduct yield of chicken processing is smaller than that for beef (mass fraction of LW basis), here it is assumed that the fraction of economic value of the processing co-products is the same for chicken as for beef (10%).

Table 3-3. Description of primary animal products considered in this study (see text). Representative of yr2007; see text for sources and description. 'PPU' = primary production unit

	Dairy	Chicken (meat)	Beef	Pig	Egg	Turkey
Primary product	Milk	Grown chicken	Grown cattle	Grown pig	Egg	Grown turkey
Primary product unit (PPU)	[g milk]	[g LW]	[g LW]	[g LW]	[egg]	[g LW]
Co-products related to reproduction (1)	"spent" dairy cow	Meat of breed stock	Meat of breed stock	Meat of breed stock	Meat of layers & breed stock	Meat of breed stock
[g LW(1) / PPU]	0.020	0.003	0.10		3.8	
[kcal co-product (1) / kcal PPU]	0.030	0.002	0.09 - 0.14		0.06	
Co-products related to reproduction (2)	"extra" calves					
[g LW(2) / PPU]	0.003					
[kcal co-product (2) / kcal PPU]	0.006					
Carcass fraction of LW	(see beef)	0.85	0.61	0.75		0.77
Retail fraction of carcass		0.86	0.70	0.78		1.00
Boneless fraction of retail		0.70	0.96	0.94		0.79
Processing (slaughter) co-products	(see beef)	Fat, feather meal, other bits, egg shells, unhatched eggs, dead chicks	Specialty meats, hide, fats, blood and bone meal, casings, other bits	Fat, skin, casings, other bits	not applicable (assuming whole egg product)	see chicken

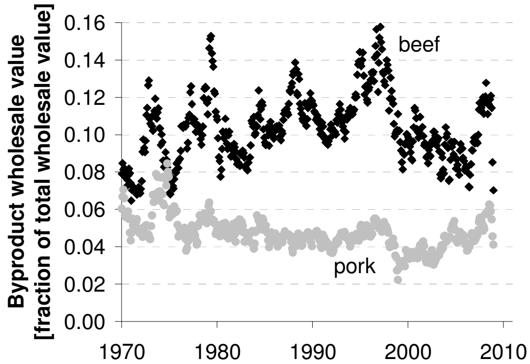

Figure 3-7. Economic value of the non-meat co-products (byproducts) of animal slaughter and processing, shown as a fraction of the wholesale value of all animal slaughter and processing products [382].

3.5.3 Estimating fodder consumption

To estimate consumption of individual feed ingredients, fodder mixes were defined for each animal category (**Table 3-4**). For non-cattle AP, specific feed consumption is reported by the USDA in its Agricultural Statistics Annual [43], in units of corn equivalent (CE) mass feed per product (**Table 3-5**). The CE feeding values of individual feed ingredients from the USDA [383] and NRC [46] are used to convert CE mass to actual mass of the assumed feed mix (see **Appendix Table 3-26**, **§3.10**). I assume that eggs produced for hatching require 1.5 times the feed of eggs produced for human consumption (g fodder DM per egg basis) to account for losses during incubation.

For cattle-based AP, specific feed consumption is calculated using estimates of feed consumption by the US EPA and total AP from USDA. The total feed consumption is reported by the US EPA for ten cattle categories in units of gross energy intake (MJ/ yr) [384]. Cattle feed mixes are used to convert the total fodder consumption calculated by the EPA to mass of individual feed ingredients for the cattle categories (**Table 3-4 (b)**).

For the purpose of showing results, I have grouped corn and sorghum silage together with other silage and green-chop (often grass and/or alfalfa). This category of fodder is referred to simply as "green". **Figure 3-8** shows the data flow of the method used to estimate consumption of fodder ingredients by animals.

Feed consumption estimates for the major AP categories considered in this work are shown in **Table 3-6** for yr2007. The livestock producing dairy, beef, poultry meat, eggs and pigs (and co-products) have consumed 95% or more of the feed (corn equivalent feeding basis) since 1990. The AP categories shown in **Table 3-6** may include more than one distinct production phase or type of animal. For example, "Dairy" includes fodder consumed by milking cows, as well as by replacement heifers that are growing and will eventually be milked. Beef cattle fodder consumption is shown separately for cattle on and outside the feedlot because of the very different fodder consumed by animals in these systems (a mostly grain and concentrate feed vs. grazed biomass, respectively).

Total concentrate fodder use in the US is also reported by USDA in its Feed Grains Database [214]. When aggregated, the feed consumption estimates developed in this work are consistent with those from Feed Grains Database. In addition, the estimates of harvested roughage fodder (**Appendix, §3.10**) are consistent with USDA estimates of total harvested roughage, as is the estimate of grazed fodder consumed by cattle. These consistency checks offer an indication that the feed mixes used here are representative of the national animal feeding situation.

Table 3-4. Constituents of fodder mix for (a) non-cattle animals (which do not consume hay, green or pasture) and (b) cattle categories. The makeup of the "concentrate" feed category for cattle and other details are shown in Appendix, §3.10. Units are [mass fraction of total fodder consumed, DM basis].

(a)	Corn (grain)	Soybean meal	Other grains	Corn gluten and wheat bits	Other co-products	Animal bits
Chicken (meat)	0.76	0.15	0.04	0.03	0.00	0.01
Pig	0.76	0.15	0.04	0.03	0.00	0.01
Egg	0.76	0.15	0.04	0.03	0.00	0.01
Turkey	0.78	0.16	0.05	0.00	0.00	0.01

(b)	Concen-trates	Non-alfalfa hay	Alfalfa Hay	Green	Pasture
Dairy cow	0.25	0.20	0.25	0.25	0.05
Dairy replacement	0.15	0.35	0.15	0.25	0.1
Bull: dairy	0.15	0.35	0.15	0.25	0.1
Beef cow	0.02	0.03	0.10	0.05	0.8
Beef replacement	0.02	0.03	0.10	0.05	0.8
Bull: beef	0.10	0.55	0.05	0.10	0.2
Beef stocker/grower	0.20	0.20	0.20	0.20	0.2
Beef on feed	0.65	0.30	0.05	0.00	0.0

Table 3-5. Specific feed consumption for non-cattle animal production [43]. 'CE'=corn equivalent; 'LW' = live weight.

	Broilers	Old chickens	Pig	Eggs (for human consumption)	Turkey
	[g feed (CE) / g LW]	[g feed (CE) / g LW]	[g feed (CE) / g LW]	[kg feed (CE) / egg]	[g feed (CE) / g LW]
1990	2.6	7.9	6.2	23	4.3
1991	2.5	7.9	6.2	22	4.3
1992	2.6	8.2	6.0	23	4.3
1993	2.4	7.9	5.9	22	4.1
1994	2.6	8.4	6.3	24	4.3
1995	2.3	8.3	5.5	20	3.9
1996	2.4	9.8	6.2	23	4.0
1997	2.4	9.8	6.4	23	3.8
1998	2.3	9.5	5.9	23	3.6
1999	2.3	8.8	5.9	23	3.8
2000	2.3	8.7	5.9	24	3.8
2001	2.3	8.9	5.9	23	3.7
2002	2.2	8.5	5.7	23	3.5
2003	2.3	9.2	5.8	23	3.4
2004	2.3	10.0	6.1	24	3.5
2005	2.2	9.4	5.9	23	3.6
2006	2.0	9.8	5.6	22	3.4
2007	2.1	10.4	5.7	22	3.5

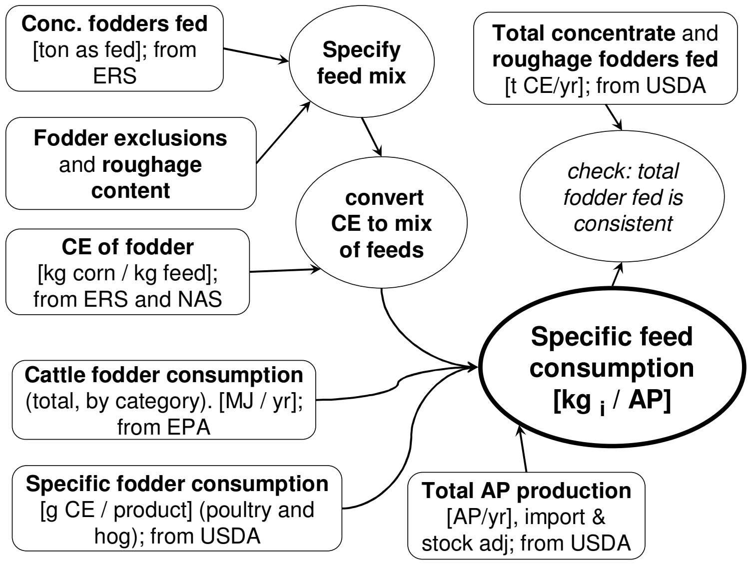

Figure 3-8. Outline of the data flow used to estimate the feed consumption associated with producing animal products in the US, as described in text. Squares represent data sources (or assumptions), while circles represent a calculation. 'AP'=animal product; 'CE' = corn equivalent. Specific feed consumption is reported for individual feed ingredients (shown in Table 3-4), as indicated by the subscript 'i'.

Table 3-6. Estimate of fodder consumption in yr2007, developed for this work (see Appendix for details). Rows correspond to AP categories; columns correspond to feed ingredients. For beef production: consumption of fodder on and off the feedlot is shown separately. For feeds: "Other co-products" includes rice, canola, cottonseed and alfalfa meals; "All silage and greenchop" includes corn and sorghum silage as well as grass and alfalfa greenchop, haylage and silage. Units are [Mt DM/ yr].

	Corn (grain)	Soy-bean meal	Other grains	Corn gluten and wheat bits	Other co-products	Animal bits	Hay, not alfalfa	Hay, Alfalfa	All silage + green-chop	Pasture
Dairy	11.5	2.3	1.5	1.6	2.4	0.0	18.4	19.3	20.6	5.0
Chicken (meat)	31.6	6.4	1.8	1.4	0.0	0.6	0.0	0.0	0.0	0.0
Beef (not on feedlot)	5.8	1.2	0.7	0.8	1.2	0.0	13.9	21.9	14.4	132.5
Beef (on feedlot)	16.7	3.4	2.1	2.3	3.5	0.0	13.0	2.2	0.0	0.0
Pig	42.0	8.5	2.4	1.9	0.0	0.8	0.0	0.0	0.0	0.0
Egg	10.2	2.1	0.6	0.4	0.0	0.2	0.0	0.0	0.0	0.0
Turkey	6.8	1.4	0.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0

3.5.4 Crop production

Crop production and yields are from USDA [43]. Crop production under irrigation for years 1992, '97, '02 and '07 are from the Agricultural Census [385]; irrigated crop production data for years 1998, '03 and '08 are from the Farm and Ranch Irrigation Survey, as is the amount of irrigation water applied to crops [386]. Yield of irrigated corn grain is shown in **Appendix** (**§3.8**, **Figure 3-13**).

The N_2O emissions for non-leguminous crops are based on the following fractions of applied N fertilizer emitted as N_2O: corn, 2.5%; wheat, 1.5%; sorghum, 1.5%; barley and oats, 2.5% [146, 147]. Other crop N_2O emissions are specified per ha, as follows: soybeans, 1.4 kg N/ ha; alfalfa, 1.3 kg N/ ha; hay, 0.6 kg N/ ha [124, 351, 387, 388]. Plausible ranges of N_2O emissions are shown in the results (**Figure 3-9**) and are described in **Table 3-20** (Appendix, **§3.8.1**).

The inputs used to produce crops are shown in **Table 3-7**, and the energy and GHG intensities of producing and delivering the inputs are shown in **Table 3-8**. Modeled GHG impacts of producing crops are shown in **Figure 3-9** and **Figure 3-10**, while **Table 3-9** shows irrigation water use for crop production in the US. Data sources and methods are as follows:

Fertilizer production & use: N application rates per land area for corn, soybeans, wheat, sorghum, barley and oats are from USDA for the following years: Corn, 2001 & 2005; soybeans, 2003 & 2006; wheat, 2000 & 2004; sorghum and barley, 2003; oats, 2005 [389]. Alfalfa nutrient requirements (kg nutrient removed/ kg alfalfa harvested) are from the University of California Alfalfa Workgroup [390]. Non-N nutrient requirements for hay are assigned to be same as corn, in terms of kg nutrient applied per tonne DM harvested; hay is assumed to be a mix of plants, including legumes, and so the N added as fertilizer is assumed to be 50% of required for corn (kg N/ tonne DM harvested). Energy requirements for fertilizer manufacture are from GREET [150] (corroborated with Meisterling et al. [158]), with N_2O emissions from nitrate-based N fertilizer production from [391]; nitrate-based fertilizers are assumed to make up 20% of N fertilizer application for all crops. Nutrient requirements per tonne of harvested DM are calculated using average yields reported by USDA [43], including years +/- one year from the years(s) for which fertilizer application is reported.

Fuels & electricity: Fuel & electricity (f&e) use for corn is from the Economic Research Service (ERS) of the USDA [389]. These include diesel and gasoline used for field operations, as well as propane, natural gas and electricity for harvested crop processing, including drying. F&e use for soy production is from Hill et al. [159]; f&e used for wheat is from Piringer and Steinberg [392]. Gasoline and diesel in oat production is from ERS [389]. Field operation fuel use for sorghum and barley grains is assumed to be equal to corn (per ha basis). Natural gas, propane and electricity use for sorghum, barley and oats is assumed to be the same as for corn (per t DM harvested basis). Fuel use for harvested roughage (per ha basis, both alfalfa and non-alfalfa) is the average of values reported in Schmer et al. [144], Adler et al. [163] and Nelson et al. ("on-site" estimate) [393] – about 1, 1, and 7 GJ/ ha, respectively. The resulting average of about 3.5 GJ/ ha is considerably larger than the 1 GJ reported in two of the three studies. However, given that many hay and greenchop fields are harvested multiple times during the year, per-ha fuel use on par with corn (which is harvested once) seems reasonable. Post-harvest natural gas use for corn and sorghum silage are ¼ that for corn and sorghum grain, respectively (per ha basis), since silage does not require drying as grain sometimes does. Propane use is assumed to be half that of grain, and electricity use per ha for silage is assumed to be equal to that for the corresponding grain crop (per ha basis). Carbon content of fuels is from EPA [244]. Upstream GHG emissions associated with the production of diesel and gasoline are from Farrell et al. [120]

and GREET [150] (10 and 23 g CO_2eq/ MJ fuel, respectively). Upstream emission from natural gas is from Jaramillo et al. [123] and propane upstream emissions are assumed to be 5 g CO_2eq/ MJ fuel. The GHG emission intensity of electricity generation in the US is regionally variable, and tends to be higher in the Midwest [394]. For this work, emissions from electricity are assumed to be 650 g CO_2eq/ kWh (delivered), which is representative of the US average.

Lime. Lime is used to maintain soil pH so that nutrients can be utilized by plants and to avoid metal toxicity (pH alters the solubility of these compounds). Lime application for corn is from Patzek [153]; soy is assumed to receive half as much lime as corn; barley and oats are assumed to receive lime at a rate the same as corn (on a kg lime/ tonne DM harvested basis); wheat, sorghum, alfalfa and hay are assumed to receive no lime. The energy used in lime manufacture is 0.6 MJ/ kg $CaCO_3$ [150]. Lime releases CO_2 to the atmosphere after application (0.44 g CO_2/ g $CaCO_3$ applied).

Seed. Seed application for corn is from ERS (USDA) [389]; seed application for soybeans is from Hill et al. [159]. Application rates for other crops are adapted from various agricultural extension reports. The energy used in seed manufacture is shown in **Table 3-7**, and is adapted from Farrell et al. [120] and Baum et al. [395]. Seeding is assumed to occur every four years for alfalfa.

Pesticide. Herbicide and insecticide use is combined in a generic “pesticide” category. Application rates for corn, soybeans, wheat, sorghum, barley and oats are from ERS [389] (as with fertilizer); application rates for alfalfa are from Mueller et al. [396]; hay is assumed to receive no pesticide. GHG impact of pesticide production is from GREET [150].

Farm machines. The annual cost of machinery (purchaser price) for corn, soybean and wheat production are from ERS [377]. These costs are converted to 2002$ using the consumer price index [76]. Energy and GHG impacts in farm machinery production (on a per 2002$ basis) are from EIO-LCA [397]. Machinery impacts for sorghum and alfalfa farming are assumed to be the same as corn (per ha basis); impacts for barley, oat and hay farming are assumed to the same as for wheat (per ha).

Other supplies. Baum et al. report that supplies not included in the above categories account for a substantial portion of overall embodied energy utilization of a diversified farm [395]. Based on these results, emissions associated with supplies are assumed to be the same as those for machinery.

Transport. The inputs listed above must be transported from where they are produced to point of use. The following transport distances are assumed: Fertilizer and pesticide are transported from production to the field 1000 km by rail (or boat) and 100 km by truck; Lime, machinery and supplies travel 500 km by rail and 50 km by truck; Seed is transported 300 km by rail and 100 km by truck. Life cycle GHG emissions of truck and rail freight transport are 180 and 30 g CO_2eq/tkm, respectively ([154-157], as described in [158]).

Table 3-7. Inputs used in crop farming. See text for references and explanation.

		Corn (grain)	Soy-bean	Wheat	Sorg-hum (grain)	Bar-ley	Oats	Alfalfa: all	Non-alfalfa: all	Corn silage	Sorg-hum silage
Fertilizers											
N	[kg / t DM]	18.7	1.8	26.4	25.8	19.9	18.8	0.0	18.7	11.0	7.0
P	[kg / t DM]	2.9	2.5	4.0	2.7	2.3	3.8	2.6	2.9	1.7	0.8
K	[kg / t DM]	6.6	8.7	3.3	0.8	2.1	7.7	20.0	6.6	3.9	0.7
Fuels											
Diesel + Gasoline	[GJ / t DM]	0.34	0.88	0.91	0.75	0.89	1.37	0.47	0.82	0.20	0.30
	([GJ / ha])	2.72	2.16	2.22	2.72	2.64	2.64	3.09	3.09	2.72	2.72
LPG, Nat gas + electricity	[GJ / t DM]	0.32	0.20	0.08	0.32	0.32	0.32	0.02	0.04	0.09	0.06
	([GJ / ha])	2.62	0.48	0.20	1.14	0.98	0.65	0.13	0.13	1.23	0.54
Lime											
$CaCO_3$	[kg / t DM]	43.6	72.2	0.0	0.0	43.6	43.6	0.0	0.0	25.6	0.0
Pesticide											
	[kg a.i. / t DM]	0.31	0.62	0.09	0.55	0.17	0.09	0.31	0.00	0.18	0.22
Seed											
	[kg seed / ha]	23	43	101	23	101	135	22	28	23	23
	[MJ for mfg. (direct) / kg seed]	9.7	20	9.7	9.7	9.7	9.7	50	9.7	9.7	9.7
	[kg CO2eq / kg seed]	0.77	1.59	0.77	0.77	0.77	0.77	3.97	0.77	0.77	0.77
Machinery											
	[$(2002) / ha]	111	124	91	111	91	91	111	91	111	111
Supplies											
	[multiple of 'Machinery'] (per ha)	1	1	1	1	1	1	1	1	1	1

Table 3-8. Energy and GHG impact associated with producing farming inputs. Sources described in text. "LC"=life cycle

Fertilizers		N	P	K
(production)	[MJ / kg]	48.4	32.0	10.6
	[g CO_2eq / MJ]	*67.3*	*73.6*	*77.5*
	[kg CO_2eq / kg]	3.3	2.4	0.8

Fuels		Diesel	Gaso-line	Propane (LPG)	Natural gas	Electricity
(production)	[MJ LC / MJ direct]	1.1	1.2	1.1	1.1	2.5
	[g CO_2eq LC / MJ direct]	79	90	65	60	172

Lime		$CaCO_3$
(production)	[MJ / kg]	8.0
	[g CO2eq / MJ]	77.3
(production & field)	[kg CO2eq / kg]	1.1

Pesticide		
(production)	[MJ / kg a.i.]	278
	[g CO_2eq / MJ]	77
	[kg CO_2eq / kg]	21

Seed		
(production)	[MJ / kg seed]	(**Table 3-7**)
	[g CO2eq / MJ]	79.3

Machinery & Supplies		
(production)	[MJ/ $]	8.28
	[g CO_2eq / MJ]	70
	[g CO_2eq / $]	576

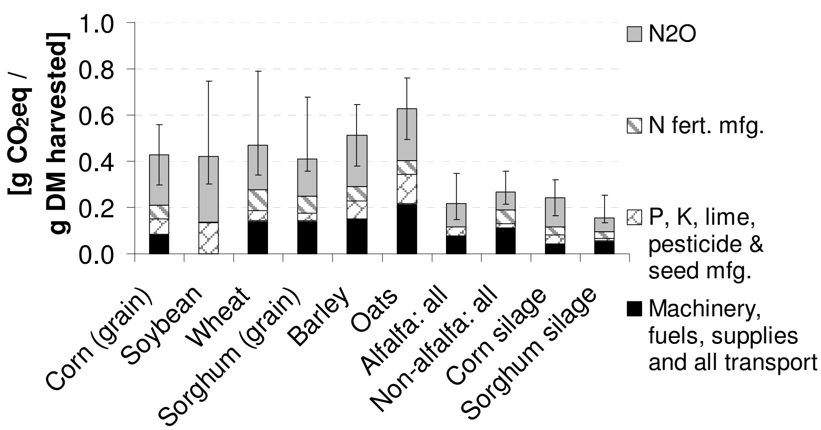

Figure 3-9. Greenhouse gas emissions associated with production of major crops in the US, with units [g CO_2eq/ g dry matter harvested]. The ranges reflect plausible N_2O emissions, shown in Table 3-20 (Appendix, §3.8.1).

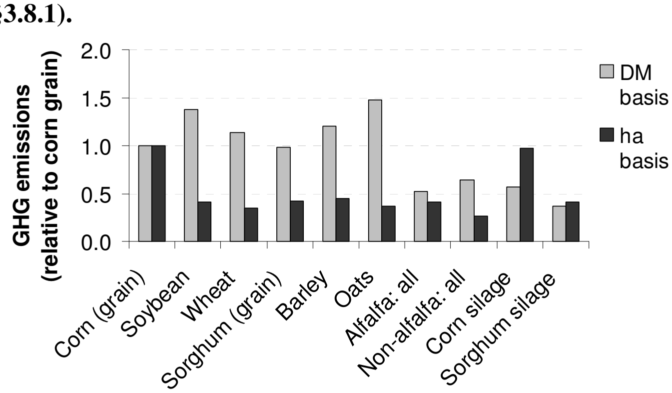

Figure 3-10. GHG impact of crops, shown relative to the impact of corn. Light grey bars represent normalized impact per DM harvested; dark grey bars show normalized impact per ha. The GHG impact of corn is higher per ha but lower per DM than the other non-roughage crops largely because of the high yield of corn.

Table 3-9. Water intensity of crop production [cu m water/ t DM harvested]. Table (a) shows water intensity of irrigated production [cu m irrigation water divided by DM produced with irrigation]; table (b) shows water intensity of all production [cu m irrigation water divided by total DM production, irrigated or not]. Note that pasture yield is not available for irrigated acres, so water intensity of pasture is shown only for "all". Calculated from data in Appendix (§3.8).

	Corn (grain)	Soybean	Wheat	Sorghum	Barley	Oats	Alfalfa: all	Non-alfalfa: all	Corn silage	Pasture
(a) irrigated										
1998	423	992	978	767	1164	2252	735	1188	372	
2003	387	868	1034	744	1023	1779	719	949	342	
2008	317	744	978	652	982	1697	834	1005	326	
(b) all										
1998	75	69	97	73	298	198	257	121	83	46
2003	67	91	110	150	350	206	253	100	104	43
2008	57	87	121	111	312	478	287	117	119	39

3.5.5 Impacts of fodder ingredients

When grains are harvested and fed whole to livestock, no allocation or system expansion is necessary if the whole crop is used as fodder. On the other hand, soybean meal and many other feeds are associated with co-products. For three of the major co-product feeds (soybean meal, wheat millfeed, and gluten feed and meal), I have derived an allocation factor based on the economic value of the feed, as a fraction of the total wholesale value of the co-products from a given parent crop. For example, gluten feed and meal are co-products of corn wet-milling (along with starch and oil). The gluten feed and meal accounts for 15% of the total value of wet-milling products, and 27% of the original DM mass of the corn grain entering the wet milling plant. Thus, one kg DM of the gluten feed and meal co-product is assigned 56% of the impact of 1 kg DM of corn grain (15% ÷ 27%). **Table 3-10** shows the allocation factors for soybean meal, wheat millfeed and corn gluten feed and meal. Animal-based feed and the 'other co-product' feed category (including alfalfa, canola, peanut and sunflower meals, along with plant fats and oils), are assigned the impact of soymeal. Other meals are assigned the impact of wheat millfeed.

All concentrate feed ingredients are assumed to be milled, requiring 290 MJ of natural gas and 140 MJ electricity per tonne of feed [398] (resulting in 40 g CO_2eq/ kg DM concentrate feed). Concentrates are assumed to travel 50 km by truck and 500 km by train to the feeding location (resulting in ~25 g CO_2eq/ kg DM feed). Hay is assumed to travel 10 km and silage to travel 5 km, both by truck. Fuel and transport GHG intensities are described in **§3.5.6.** The land, water, and GHG impacts of producing fodder ingredients are shown in **Table 3-11**.

Table 3-10. Allocation factor (last row) and derivation for crop-based co-product feeds. Soybean from [216]; wheat mill from [399]; corn from [400, 401].

		Soybean meal	Wheat mill feed	Gluten feed and meal
Parent crop	(crop or primary product)	Soybean	Wheat grain	Corn grain
Feed value	[fraction of total wholesale economic value]	0.64	0.06	0.15
Feed yield	[fraction of mass of primary product]	0.80	0.29	0.27
Allocation factor	[fraction of impact of equal mass (DM) of parent]	0.80	0.21	0.56

Table 3-11. Land, water and GHG impacts (associated with crop production) of feeds considered in this work. Feeds are grouped as (a) grains; (b) co-product concentrates; (c) roughages. Estimates are process-based, except the co-product concentrates: *italic* values are estimated using economic-based allocation; underlined values are defined by an offset marginal feed (see text). Process-based impacts of soybeans (rarely fed to livestock whole) are shown with grains to provide a basis for the soybean meal allocation. The area impact for pasture includes only "cropland pasture"; if all grazed land is considered the area increased to 13.8 sq m/ kg DM.

(a)		Corn (grain)	Wheat (grain)	Sorghum (grain)	Oats	Barley	(Soybeans, whole)
Land	[sq m / kg DM]	1.3	4.3	2.5	5.4	3.6	4.1
Water	[L / kg DM]	317	978	652	1697	982	744
GHG	[g CO2eq / kg DM]	381	391	412	542	438	528

(b)		*Soy-bean meal*	*Wheat mill feed*	*Gluten feed and meal*	Other meals (1)[a]	Other meals (2)[a]	Animal bits
Area	[sq m / kg DM]	*3.3*	*0.5*	*0.7*	3.3	0.5	3.3
Water	[L / kg DM]	*599*	*210*	*176*	599	210	599
GHG	[g CO2eq / kg DM]	*476*	*88*	*254*	476	88	476

(c)		Pas-ture	Hay, alfalfa	Hay, non-alfalfa	Green, alfalfa	Green, non-alfalfa	Corn silage	Sorg-hum silage
Area	[sq m / kg DM]	1.3	1.6	2.6	1.6	2.6	0.7	1.1
Water	[L / kg DM]	39	834	1005	834	1005	326	326
GHG	[g CO2eq / kg DM]		162	151	162	151	205	160

[a] Other meals (1) =Alfalfa meal, Canola meal, Fats and oils, Peanut meal, Rice (millfeed), Sunflowerseed meal; (2)= Cotton-seed meal, Misc by-products, Linseed meal

3.5.6 Impacts from animals & facilities

Animal farm operations and infrastructure are associated with energy and resource use, and animals are often transported between production phases and to processing facilities. **Table 3-12** shows assumed transport requirements for animals.

The estimates of GHG impacts of animal farm operations (not including fodder production or delivery) are calculated based the amount of feed delivered to animals. The GHG intensity factor for animal farms is estimated to be 120 g CO_2eq per kg DM fodder fed (adapted from Pelletier [398], who reports that on-farm inputs and emissions during chicken growing amount to 230 kg CO_2 per tonne of bird live weight produced). On-farm operations and infrastructure for dairy farms are assumed to be 50% more intensive per kg DM fodder than other farming operations, due to the milking infrastructure.

Methane is generated in the in the digestive system of ruminants, and is usually characterized as a fraction of the feed intake that is emitted as CH_4 on a gross energy intake basis. For roughage (including grazed and harvested grass and alfalfa), this fraction is often 5-8%. For ruminants on grain, the methane conversion factor is lower – usually 2-4% [402] (see **§ 3.3.1**). The methane emissions factors used in this work are adapted from US EPA and are shown in **Table 3-13**.

Methane and nitrous oxide emissions from animal excreta depend heavily on management. Methane emissions from manure managed with open lagoons and pits can be quite high, while CH_4 from manure in cattle feedlots in arid locations is low. The EPA estimates N_2O from manure using the DAYCENT model for most of the manure produced in the US. Managed manure that is spread on fields daily is treated in the "Agricultural soils" category, as is manure deposited by grazing animals on pasture, range or paddock (PRP). Emissions associated with manure management from the US EPA are shown in **Table 3-14** (see **Chapter 5** for an LCA of manure management).

Table 3-12. GHG impacts associated with animal transport. GHG impact of truck transport as in [158] (tkm = tonne km). Distances and trailer capacities are shown in Appendix (Table 3-24).

	Dairy	Chicken (meat)	Beef (not on feedlot)	Beef (on feedlot)	Pig	Egg	Turkey
(production unit)	[t milk]	[t LW]	[t LW]	[t LW]	[t LW]	[k eggs]	[t LW]
[kg CO_2eq / prod. unit]	0.8	7.1	16.7	11.9	9.8	0.2	4.6
[tkm / prod. unit]	4.7	39.4	91.9	65.6	54.3	0.8	25.3
[truck km / prod unit]	0.2	2.0	4.6	3.3	2.7	0.0	1.3

Table 3-13. Methane emissions from cattle in US, reported in fraction of gross energy intake (MJ). Adapted from US EPA (Table A-16 in ref [384]).

	Dairy cow	Dairy replace-ment	Bull: dairy	Beef cow	Beef replace-ment	Bull: beef	Beef stocker	Beef feedlot
[% of gross E intake]	0.054	0.059	0.059	0.065	0.065	0.059	0.065	0.030

Table 3-14. Total emissions (yr2007) of methane (CH_4) and nitrous oxide (N_2O) associated with manure in the US, in units of [Tg CO_2eq] [244]; GWP of CH_4 and N_2O are 25 and 298, respectively, with a time horizon of 100 years.

	Dairy	Chicken (meat)	Beef (on feedlot)	Beef (not on feedlot)	Pig	Egg	Turkey
CH_4	23.2	0.8	0.6	2.4	23.9	2.3	0.2
N_2O	3.7	1.0	6.4	0.0	1.5	0.4	0.3

3.5.7 Land use change

The United Nations report on the global impacts of livestock finds that emissions from land use change are 35% of the total GWP of livestock, which is slightly larger than the share from enteric methane emissions from ruminants [301]. Let us first consider the magnitude of land use conversion (LUC) CO_2 emissions if an animal feed does, in fact, alter land use patterns. The land requirement for the feeds in **Table 3-11** range from 0.5 to 5.4 m^2/ kg DM. Corn grain requires an average of 1.3 m^2 of land per kg DM in the US. Consider a situation where corn was planted to feed an animal on land previously covered with perennial grass, and the LUC emissions are 50 t CO_2/ ha [114]. If we allocate this "pulse" of C over 20 years in a linearly declining manner (such that emissions allocated to year 21 are zero, as in [284]), 10% of the emissions are allocated to the first year. In this representative scenario, C emission for the year following LUC amounts to 500 g CO_2/ sq m, or 650 g CO_2eq per kg DM of corn grain. Alternatively, if the emissions pulse from LUC is allocated uniformly over 30 years, emissions amount to 220 g CO_2eq/ kg DM of corn grain. The GHG impact of corn without any LUC was calculated to be about 380 g CO_2eq/ kg DM (**Table 3-11**), so these magnitudes of LUC emissions are substantial. On the other hand, if LUC emissions are allocated over hundreds of years, the impact of LUC emissions per kg DM harvested is minimal.

3.6 Results: Life cycle impacts of animal farming in the US

The life cycle GHG impacts of AP in the US are shown in **Figure 3-11**, while **Table 3-15** includes water and land impacts. The total GHG impact from the animals considered here totals 360 Tg CO_2eq per year (around yr2007). Considering the total (gross) GHG emissions of 7150 Tg CO_2eq in the US for yr2007, the emissions associated with producing the animals considered in this work amount to about 5% of gross US emissions.

For beef, poultry and pig meat, any co-products related to the reproductive cycle are aggregated with the primary product, so slaughter byproducts are the only co-products shown in **Table 3-15** for these types of animals. For diary and eggs, the net impacts (after credits for meat co-products related to the reproductive cycles) are 94%-97% of the gross impacts. These results will be used to assess integrated animal and fuel production systems and changing biomass utilization (**Chapter 6**).

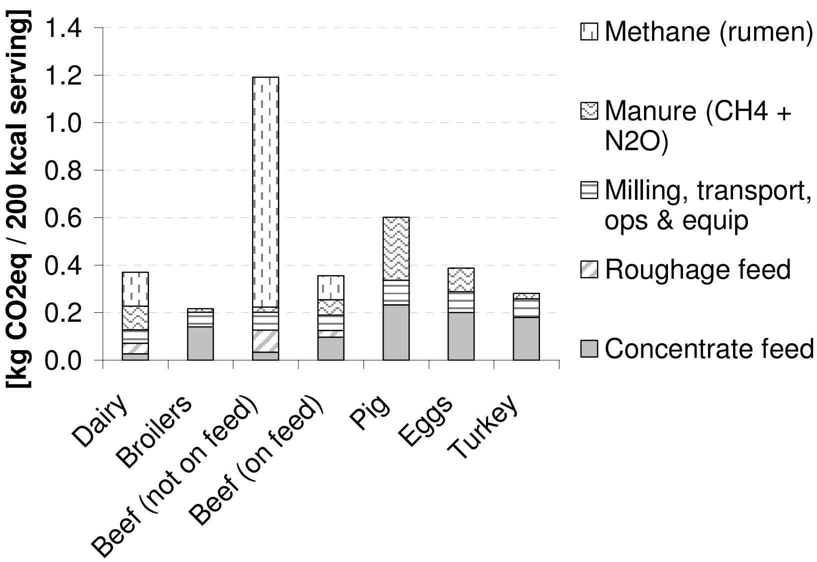

Figure 3-11. GWP of animal products, before allocation and system expansion. Results for beef are differentiated between cattle on the range and cattle on feed (thus, the total impact for beef is the sum of the two beef categories shown).

Table 3-15. GWP, water and land impact associated with animal products. "NET" represents impacts after allocation to processing (slaughter) co-products (where applicable) and applying credit for co-products related to the animals' reproductive cycles (in the case of Dairy and Eggs).

		Dairy	Chicken meat	Cattle meat	Pig meat	Eggs	Turkey meat
NET GWP impact	[g CO2eq / 200 kcal serving]	349	210	1325	573	387	237
NET land impact	[sq m / 200 kcal serving]	0.48	0.51	2.96	0.88	0.76	0.65
NET water impact	[L irrigation / 200 kcal serving]	187	118	607	205	178	150
Primary product	(unit)	[g milk]	[g LW]	[g LW]	[g LW]	[egg]	[g LW]
200 kcal serving	[(primary product unit / 200 kcal serving]	312	185	173	140	3	171
Production in US	[G 200 kcal servings / yr]	270	121	109	96	27	20
	[Tg CO2eq / yr]	99	28	160	58	11	5
Total GWP impact	[g CO2eq / primary product]	1.17	1.26	8.53	4.35	0.13	1.54
	[g CO2eq / 200 kcal serving]	365	233	1473	610	400	264
Total land impact	[sq m / 200 kcal serving]	0.51	0.56	3.29	0.94	0.80	0.72
Total water impact	[L irrigation / 200 kcal serving]	195	131	674	218	185	167
Processing co-product allocation	[fraction of total impact allocated to slaughter byproducts]	(na)	0.10	0.10	0.06	(na)	0.10
Reproductive cycle co-product(s) credit	[g CO2eq / 200 kcal serving]	16.1				13.2	
	[sq m / 200 kcal serving]	0.027				0.032	
	[L irrigation / 200 kcal serving]	8.10				7.36	

3.7 Comparison and discussion

In a review of GHG mitigation options in the animal sector, Monteny et al. [306] indicate the following as promising reduction opportunities: improved animal productivity, feeding more starch to ruminants, more efficient N fertilizer use, and better manure heap management (adding C). Gill and Smith also review GHG emissions associated with livestock globally and the lifestyle changes, farming practices and technologies that can reduce the impact of livestock. Developing effective policies to reduce GHGs from animal agriculture is difficult, however. Three principal reasons are uncertainties related to the GHG impacts of livestock production (in particular, land use and N_2O emissions), cultural aspects of AP consumption, and the connection between agricultural commodity prices and food security. In addition, efficiency of animal production can be measured in terms of human edible feed input. When viewed this way, ruminants can increase food security in regions where roughages (not edible by humans) are available [403].

Elferink [404] reports that broilers, pork and beef need 0.56, 0.56 and 1.33 sq m per 200 kcal serving (converted from square meters per kg of meat with the maize in Denmark scenario), while Eriksson et al. compare pig growing based on different feeds, and find impacts between 180 and 210 g CO_2eq per 200 kcal serving (1.3-1.5 g CO_2eq/ g LW gain) [405].

Basset-Mens and van der Werf assess the environmental impact of three pork producing scenarios in France [406]. Pig production is associated with 320-560 g CO_2eq (2.3, 3.5 and 4.0 kg CO_2eq per kg LW for the “Good agriculture practices,” “red label” (a quality standard) and “organic agriculture” scenarios in France). Feed production contributes most uncertainty in the LCA results, while N_2O emissions from housing can be substantial for systems using straw bedding as opposed to slatted floors. If food processing residue is used as feed, impacts of meat production are reduced, but allocation procedure affects these results [407].

Pelletier conducts a life cycle assessment of US broiler chicken production [398]. Results indicate a net of 260 g CO_2 per 200 kcal serving of meat (1.4 kg CO_2eq/ kg LW). More than 80% of the gross impacts are attributable to the feed. Of the feed impact, about 40% is attributable to corn, and 45% is attributable to poultry processing byproducts (29 and 17% for “poultry meal” and “poultry fat”, respectively).

The feed consumption calculated in this work is representative of the national average. Such an average should be looked at with skepticism, since, for example, cattle in Montana will likely have different feed requirements and feeds available than in Tennessee. However, I believe this approach offers a useful mix of simplicity and specificity regarding feed ingredients, such that the results can be scrutinized and adjusted if needed to represent a particular case which is materially different from the aggregates used here, while offering easily interpretable results of biomass appropriation of the various APs.

Perennial grasses might be a preferred feedstock for cellulosic ethanol production. As **Figure 3-12** shows, the beef cow-calf system to a large extent relies on grazed biomass for fodder. If cellulosic ethanol using perennial grasses for feedstock are developed at a large scale, such a demand shock for cellulosic feedstock could have substantial impacts on the beef industry. In particular, some animal protein production could be shifted to other, less biomass intensive AP (e.g. chicken and herbivorous fish). Since methane from cattle is a major GHG impact of AP, this possible change could be associated with lower GHG emissions from animal agriculture (see Discussion, **Chapter 6**).

Alternatively, the microbial processes that occur in the rumen of livestock could be altered to provide the same service to the animal (e.g. breaking up cellulose), but without producing methane. Fonty et al. demonstrated that "reductive acetogenesis" can replace methanogenesis to provide methanogen-free rumen function in lambs [408]. Such a breakthrough may reduce GHG emissions from AP, but will not necessarily improve feeding efficiency.

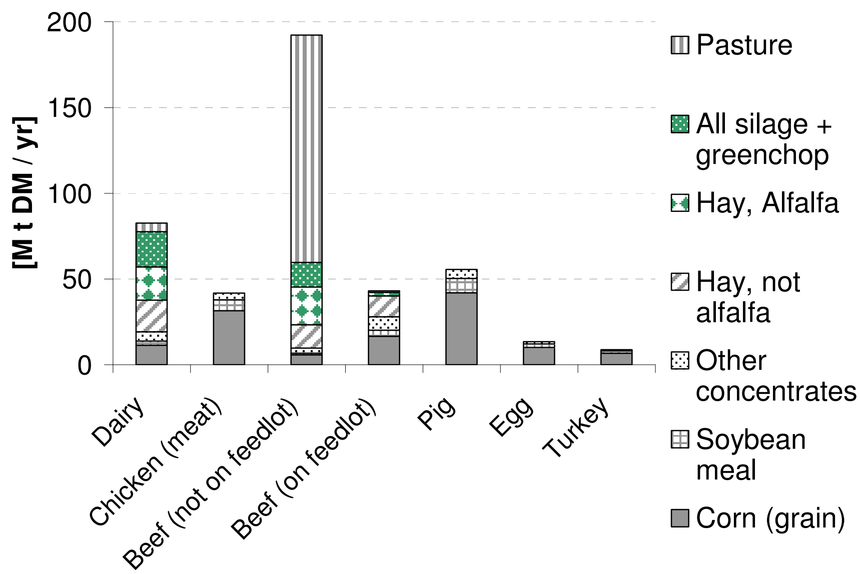

Figure 3-12. Total fodder consumed by livestock in US, yr2007. Data from Table 3-6.

3.8 Appendix 1: Crop production

Table 3-16 shows the total dry matter (DM) production of major crops in the US. **Table 3-17** shows yields for (a) all crops and (b) irrigated land. **Table 3-18** shows the fraction of crops produced under irrigation, while **Table 3-19** shows water application rates.

Table 3-16. Total dry matter (DM) production in US. Grains, corn silage and sorghum silage from [43]; harvested roughage is adapted from [43, 385, 409-411] and includes hay and grass and alfalfa greenchop (see §3.10 for details). Converted from standard moisture to DM using factors in Table 3-25.

	Corn (grain)	Soy-bean	Wheat	Sor-ghum	Bar-ley	Oats	Harvested roughage	Corn silage	Sorghum silage	Pasture
1990	170	46	64	13	8	4	121	28	2	197
1991	160	47	47	13	9	3	126	26	1	201
1992	203	52	58	19	9	4	121	28	2	200
1993	136	44	56	12	7	3	121	26	1	210
1994	216	60	55	14	7	3	124	29	1	212
1995	159	51	51	10	7	2	127	25	1	214
1996	198	56	54	18	7	2	124	27	1	206
1997	198	64	58	14	7	2	126	31	2	199
1998	209	65	60	11	7	2	125	30	1	199
1999	202	63	54	13	5	2	132	30	1	191
2000	213	65	52	10	6	2	127	32	1	183
2001	204	68	46	11	5	1	129	32	1	184
2002	192	65	38	8	4	1	124	32	1	186
2003	217	58	55	9	5	2	130	34	1	178
2004	253	74	51	10	5	1	131	34	1	179
2005	239	73	50	9	4	1	124	34	1	182
2006	226	76	43	6	3	1	116	33	1	187
2007	280	63	48	11	4	1	121	34	2	189
2008	260	70	59	10	5	1	121	35	2	183

Table 3-17. Crop yields [t DM / ha]. Table (a) shows average yield of all harvested acres [43]; table (b) shows average yield of irrigated acres [385, 386, 409-413].

(a)

	Corn (grain)	Soy-bean	Wheat	Sor-ghum	Barley	Oats	Alf hay	Non-alf hay	Alf green	Non-alf green	Corn silage
1990	6.3	2.0	2.3	3.4	2.6	1.9	6.4	3.4	8.5	4.0	11.1
1991	5.8	2.0	2.0	3.2	2.6	1.6	6.4	3.7	8.5	4.3	10.4
1992	7.0	2.2	2.3	4.0	2.9	2.0	6.4	3.8	8.5	4.4	11.3
1993	5.3	1.9	2.2	3.3	2.7	1.7	6.3	3.7	7.4	4.8	9.3
1994	7.4	2.4	2.2	4.0	2.6	1.8	6.6	3.9	7.6	5.0	12.4
1995	6.0	2.1	2.1	3.0	2.7	1.7	6.7	3.9	7.8	5.0	11.5
1996	6.7	2.2	2.1	3.7	2.7	1.8	6.4	3.7	7.4	4.8	12.1
1997	6.7	2.3	2.3	3.8	2.7	1.8	6.5	3.8	7.6	4.9	12.6
1998	7.1	2.3	2.5	3.7	2.8	1.9	6.8	3.7	6.9	4.8	12.7
1999	7.1	2.1	2.5	3.8	2.8	1.8	6.8	3.7	7.0	4.8	12.4
2000	7.3	2.2	2.4	3.3	2.8	2.0	6.8	3.8	6.9	4.9	13.2
2001	7.3	2.3	2.3	3.3	2.7	1.9	6.5	3.7	6.7	4.8	13.0
2002	6.9	2.2	2.0	2.8	2.6	1.7	6.2	3.6	6.3	4.7	11.3
2003	7.5	2.0	2.6	2.9	2.7	2.0	6.3	4.0	6.0	4.3	12.8
2004	8.5	2.5	2.5	3.8	3.2	2.0	6.8	4.0	6.4	4.4	13.8
2005	7.8	2.5	2.4	3.7	3.0	2.0	6.6	3.7	6.3	4.1	14.1
2006	7.9	2.5	2.2	3.1	2.8	1.9	6.5	3.5	6.2	3.8	12.7
2007	8.0	2.4	2.3	4.0	2.8	1.9	6.5	3.8	6.1	4.1	13.8
2008	8.2	2.3	2.6	3.5	3.0	2.0	6.5	3.8	6.1	4.1	14.7
2009	8.6	2.5	2.6	3.5	3.4	2.1					

(b)

	Corn (grain)	Soy-bean	Wheat	Sor-ghum	Bar-ley	Oats	Alf: all	Non-alf: all	Alf hay	Non-alf hay	Alf green	Non-alf green	Corn silage
1992	7.7	2.2	4.0	4.0	3.9	2.3			8.6				16.2
⋮													
1997	8.3	2.4	4.4	4.4	4.2	2.3			9.2				17.6
1998	8.6	2.5	4.4	4.4	4.2	2.3	9.9	4.9					18.0
⋮													
2002	8.4	2.5	4.2	3.6	4.1	2.5			9.2	4.4	7.6	10.3	17.5
2003	9.4	2.8	4.4	4.1	4.5	2.4	9.8	5.5					19.6
⋮													
2007	9.5	2.7	4.7	4.7	4.7	2.6			9.6	4.9	7.3	11.3	19.5
2008	9.6	2.9	4.4	4.2	4.7	2.5	8.8	5.5					19.6

Table 3-18. Fraction of total production (DM basis) under irrigation in US (total production data from Table 3-16, while irrigated production is calculated from [385, 386, 409-413]). 'Alf'= alfalfa.

	Corn (grain)	Soy-bean	Wheat	Sor-ghum	Bar-ley	Oats	Alf hay	Non-alf hay	Alf green	Non-alf green	Corn silage
1992	0.14	0.04	0.10	0.14	0.20	0.04	0.31				0.20
⋮											
1997	0.17	0.06	0.10	0.10	0.25	0.05	0.35				0.23
1998	0.18	0.07	0.10	0.09	0.26	0.09	0.35	0.10			0.22
⋮											
2002	0.16	0.08	0.11	0.12	0.33	0.06	0.42	0.09	0.12	0.36	0.31
2003	0.17	0.10	0.11	0.20	0.34	0.11	0.35	0.11			0.31
⋮											
2007	0.17	0.08	0.10	0.14	0.32	0.07	0.44	0.11	0.13	0.49	0.34
2008	0.18	0.12	0.12	0.17	0.32	0.27	0.34	0.12			0.37

Table 3-19. Water applied to irrigated acres. Units= [m^3 / ha] (data from [386, 412, 413]).

	Corn (grain)	Soy-bean	Wheat	Sor-ghum	Barley	Oats	Alf: all	Non-alf: all	Corn silage	Pasture
1998	3658	2438	4267	3353	4877	5182	7315	5791	6706	5791
2003	3658	2438	4572	3048	4572	4267	7010	5182	6706	5182
2008	3048	2134	4267	2743	4572	4267	7315	5486	6401	4877

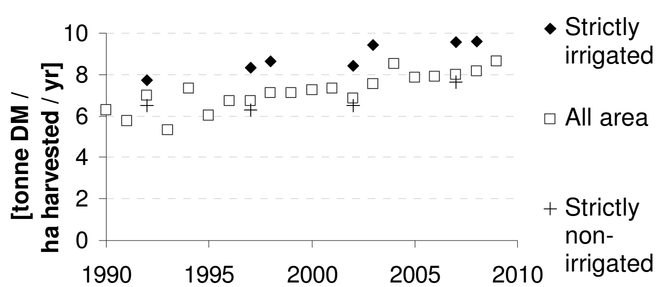

Figure 3-13. Comparison of irrigated yields with average yield and non-irrigated yield for corn grain. Calculated with data from tables above.

3.8.1 N_2O emissions

Table 3-20. Nitrous oxide (N_2O) emissions from crop production. (a) Fraction of N applied that is eventually emitted as N_2O, including emissions direct from the field, and emissions from that leaves the field via runoff, leaching or volatilization. Corn, oats and barley are assigned baseline emission factors of 2.5% [146, 147]. (b) kg N / ha. Values are calculated from the parameters in (a) and the nitrogen application rates in Table 3-7, except for the underlined categories (soybean, alfalfa and non-alfalfa roughage). These underlined categories, which include leguminous nitrogen-fixing plants, are specified from [124, 351, 387, 388].

	Corn (grain)	Soybean	Wheat	Sorghum (grain)	Barley	Oats	Alfalfa: all	Non-alfalfa: all	Corn silage	Sorghum silage
(a) fraction of applied N										
low	0.01		0.005	0.01	0.01	0.01			0.01	0.01
mid	0.025		0.015	0.015	0.025	0.025			0.025	0.015
high	0.04		0.04	0.04	0.04	0.04			0.04	0.04
(b) kg N / ha										
low	1.42	0.80	0.33	0.74	0.54	0.37	0.40	0.20	1.42	0.74
mid	3.56	1.40	0.98	1.11	1.36	0.92	1.30	0.60	3.56	1.11
high	5.70	3.00	2.61	2.97	2.18	1.47	3.00	1.30	5.70	2.97

3.9 *Appendix 2: Animal product production in the US*

To a large extent, dairy and beef production are distinct operational processes in the US. Specialized cattle breeds are used by dairies to maximize milk production. Likewise, beef cattle breeding focuses on producing animals which efficiently convert feed to meat under certain climates and feeding regimes. In both dairy and beef production, some of the calves produced are kept as “cow replacements”; these calves eventually replace a cow that is “retired” (often for slaughter).

Beef production

Beef calf production in the US is characterized as a grazing system, meaning that much of the nourishment required to feed the pregnant and lactating beef cows comes from grazed biomass. Gestation time is about 9 months, and beef cattle are weaned at about 500 lb (530 lb is the average [414], but environmental and market conditions drive variations). About 15% of calves stay on the operation to become a beef cow [414]. A small number of calves are slaughtered for meat. Most calves enter a “stocker/replacement” phase, from which they may enter a feedlot.

In a feedlot, cattle consume a ration with high energy content, typically corn-based, with other low-cost ingredients and supplements necessary for good weight-gain performance. Assuming cattle spend 160 days on a feedlot [415], about 95% of slaughtered cattle in the years 2005-2008 were from feedlots (slaughter and inventory data from [43]).

Veal is beef from a slaughtered calf (as opposed to a grown adult). Since yr1999, veal has made up less than 1% of total beef production. Thus, veal is not considered as an independent product in this work. However, veal may be an important co-product of dairy production, since the number of calves slaughtered annually for veal amount to about 25% of the "extra" calves from dairy production [43].

Dairy production

Milk production in the US is characterized by relatively large confinement operations and involves not only feeding the milking cows, but also producing "replacement" dairy cows. In order for a dairy cow to produce milk, it must have given birth. Thus, milk production involves harvesting the milk produced by a cow after she calves. The cow and calf are separated almost immediately after birth, and milking the cow begins. In addition, dairy production requires manure managing, feeding equipment (e.g. tractors to process and deliver feed), milking equipment (on the farm), milk cooling, milk collections, and milk processing and bottling. Bottled milk is then delivered to retail, where milk is kept cool until purchase, after which milk is, again, cooled. An average dairy cow in the US produces over 9 t milk per year (**Table 3-21**).

The cow is impregnated 3 or 4 months after giving birth (while milking), and milking ceases during a "dry period" a couple months before birth. A milking period usually lasts for 10 months. Calving cycles in the US average 13.2 months [381]. Eventually, dairy cows must be replaced, either because of death or "retirement". Thus, dairy production involves also raising heifers to replace cows. The ratio of replacement heifers to milking cows is 0.45, as of yr2008 [43]. Most retired or "spent" dairy cows are sold for slaughter. An average dairy cow in the US is milked for 3 years before being slaughtered (calculated assuming a dairy cow death rate (not used for meat) of 5.7% [381], and dairy cow slaughter data from [43]). Slaughtered dairy cows (which may spend some time on a feedlot before slaughter) have thus made up 7%-8% of total cattle slaughter live weight (LW) from 1999 through 2008.

Not all calves produced by dairy cows are needed as cow replacements. The "extra" calves are sold for beef production and are a co-product of dairy production. Based on an average cow that produces 2.5 live-birthed calves over its productive life, and that 1.2 live-birthed calves are needed to produce one "new" milking cow (due to deaths), 40%-50% of dairy calves are required to eventually replace dairy cows that are "retired" or that die. So, each cow produces (2.5 – 1.2) 1.3 "extra" calves over its reproductive life.

Decisions made about when to retire a milking cow have many factors, including expectations about relative prices of milk and meat, as well as cow health and management. Due to these considerations, the practices on a particular dairy could be different from the averages presented here. However, the averages will be used to inform a baseline LCA, particularly to calculate impacts related to the co-products of "spent" cows and "extra" calves. See **§3.5.2** for further discussion of animal co-products.

Cows having four or fewer lactation cycles before retirement or death account for 75% of the total milking herd. The number of dairy cows slaughtered annually amount to between 25% and 30% of the dairy cow population (average=29%; SD=2.1%) [43, 416].The cow death rate (different than permanent removals for slaughter) is 5.7% [417]. The resulting 34.5% annual replacement rate implies an average milking life of 2.9 years (duration between first calving and "retirement" or death). This estimate is corroborated by a US EPA reported average of 2.5 milking cycles / cow, which corresponds to a milking life of 2.8 years using a lactation cycle of 13.2 months [381].

Table 3-21. Annual milk production per dairy cow in US [43].

		2000	2001	2002	2003	2004	2005	2006	2007	2008
Milk production	[t milk / cow / yr]	8.3	8.2	8.4	8.5	8.6	8.9	9.0	9.2	9.3

Trade

Trade in cattle-related goods includes live cattle as well as meat. The US has consistently been a net importer of beef (carcass-equivalent basis). Net imports have typically been around 2% - 5% of total production (since yr1993), with some years much higher when BSE (mad cow) found in the US herd restricted exports [378].

In 2009, net milk exports amounted to 2.5% of production, although, the US is a net milk importer over the period between 1990 and 2009 (0.3% of production, SD=1.9%). Trade of dairy cows and heifers is not considered [378].

Pork and Poultry-related production

To produce poultry meat, breeding stock first lay eggs, which then are hatched in nurseries. The resulting chicks are typically transported from the chick production facility to a growing operation. When the grown birds are ready for market (often within 50 days for chickens [415]), they are collected and transported to a killing and processing facility. Chickens grown specifically for meat (often termed "broilers") have made up at least 97% of all chicken slaughter (LW basis) since yr1990. The remainder of chicken meat production is from breeder stock – the layers which produce eggs for broiler hatching, as well as for table egg consumption.

Of all eggs produced by laying hens in the US, 85% are used for human consumption, 14% are used for broiler chick production and 1% is used for egg-layer replacement [380]. These ratios were used to estimate the fraction of breeder stock chicken slaughter from the table egg and chicken meat systems. Using the average layer population and table egg production from NASS [380], hens produce about 270 eggs per year.

Turkeys average 29 lb / head [43] and about 120 days old [415] at slaughter. About 10% of growing turkeys die before market weight [43].

Pork production entails producing weaned piglets and growing them to a market weight and can be considered to occur in three phases. The first phase involves producing weaned piglets, whereby sows are impregnated, give birth, and suckle piglets for 20-35 days [415, 418]. Gestation time is 115 days and sows generally produce 9-10 weaned piglets per pregnancy [415]. Piglets spend 45 days in the nursery, and are slaughtered when they are about 180 days old [415, 418]. The weaned piglets then enter a "nursery" phase where they grow for an average of 47 days [418]. In the nursery, heaters may be used to maintain healthy living conditions for the piglets. During the "finishing" phase pigs grow to market weight. These distinct phases may all occur at one location, or at different locations, with animals transported to and from facilities.

The life span of a sow (reproducing female pig) is less than two years (based on the ratio of annual sow slaughters and average sow population from NASS). Pigs slaughtered from the breeding population (boars and sows) have made up 5% of slaughter weight over the years 2003 through 2008.

Table 3-22. Production levels of reproductive co-products related to dairy and egg production.

	Dairy		Eggs for eating (table)
	[Mt LW from spent cows / yr]	[Mt LW from 'extra' calves / yr]	[Mt LW chicken from 'spent' breeders]
1990	1.3	0.3	0.3
1991	1.5	0.3	0.3
1992	1.5	0.3	0.4
1993	1.5	0.3	0.3
1994	1.5	0.3	0.4
1995	1.5	0.3	0.4
1996	1.6	0.3	0.1
1997	1.5	0.3	0.2
1998	1.4	0.3	0.1
1999	1.4	0.3	0.2
2000	1.4	0.3	0.2
2001	1.4	0.3	0.3
2002	1.5	0.3	0.2
2003	1.6	0.3	0.4
2004	1.3	0.3	0.4
2005	1.3	0.3	0.2
2006	1.3	0.3	0.1
2007	1.4	0.3	0.3
2008	1.5	0.3	0.3

3.9.1 Protein and calorie AP supply in the US

In **Figure 3-6** (**§3.2**), the total domestic supply of for beef, pork, dairy and eggs from 1950-present; chicken and turkey meat protein is estimated from 1960-present (data not available before 1960). Protein content of each product is adapted from USDA [376] and Smil [33] (**Table 3-1**). Product-specific data sources and assumptions are as follows:

Eggs

- Edible mass of an egg is 47g (average of medium and large eggs) [376]
- Domestic supply for years 1993-2008 are from ERS Poultry Outlook [419] ("domestic disappearance").

- For years 1960-1992 I estimate domestic use as follows: total production and trade data from Poultry Yearbook [379]; fraction of total egg production used for hatchings is average of 1993-2008 (14.9 %, S.D.=3.0%) from Poultry Outlook.
- Years 1950-1959 are the same as 1960-1992, except trade data is not available, so I assume net exports amount to 0.2% of total production (average of years 1960-1969).

Dairy

- Domestic production for years 1950-2008 from NASS [43].
- Domestic supply of dairy products for human consumption for years 1970-2008 from ERS Livestock, Dairy and Poultry Outlook, supporting tables [378]. To estimate domestic supply as a fraction of total production for years 1950-1969, I apply the average factor for 1970-2008 (99% of production; SD=2%).

Meat (general)

- Annual domestic slaughter data for beef and pork are available for the years 1950-present, while data are available for chicken and turkey production from 1960-present, from NASS [43] and/or ERS Livestock, Dairy and Poultry Outlook tables [378]. NASS data are available in live weight; ERS data are typically dressed weight.
- Conversions between live and dressed weight are made using data from NASS which report federally inspected live and dressed weights for beef and pork (1950 to present), and total live and chilled (“ready to cook”) weights for chicken and turkey (1960-present). (note: ERS also reports carcass/live weight ratios for beef and pork and “ready to cook” : live weight ratios for chicken and turkey [377, 378])
- Trade, supply and disappearance data are available from 1970-2004 from the Red Meat Yearbook [377] and from 1993-present from the Livestock, Dairy and Poultry Outlook tables [378]. I have estimated domestic supply from 1950-1969 for beef and pork (see below)
- Boneless weight is assumed to represent the edible fraction (and thus food available to people) of the primary animal product (see **§3.5.2** for details on co-products)

Chicken and Turkey

- Data is available only since 1960
- Since trade data is not available for years 1960-1969, I assume domestic disappearance is 97% of production (average of 1970-1990 from [379])
- Conversions from carcass weight to boneless (edible) weight are from [377] for years 1970-2004; years 2005-present are estimated from the same data using the average for 1994-2004 (0.60); years 1960-1969 use conversion of 0.70 from [378].

Beef and Pork

- Domestic disappearance (in carcass weight) for years 1970-1992 are from [377]; years 1993-present are from [378]
- To estimate domestic supply from domestic production for years 1950-1969, I apply the average Supply/Production factor from 1970-2008 (For beef: 1.06, S.D.=0.03; for pork: 1.01, S.D.=0.03).

- Boneless:Carcass weight ratios are available from 1970-2004 from [377]. The ERS Outlook tables [378] assume a constant factor of 0.67, and I apply this factor for years 2005-present. For years 1950-1969, I apply a factor of 0.70, equal to the value from 1970 [377].

Table 3-23. Protein content of animal products. Top row shows protein content as mass fraction of the edible animal product (moist, as-eaten basis). The middle row reports the DM fraction of the animal product (or, 1 – water mass fraction). Bottom row reports the mass fraction of dry matter (DM) that is protein.

	Milk (3.25% fat)	Chicken	Beef	Pork	Turkey	Egg
[g protein/ g edible]	0.035	0.200	0.174	0.140	0.181	0.130
[g DM/ g edible]	0.117	0.381	0.423	0.502	0.322	0.242
[g protein/ g DM]	0.276	0.488	0.412	0.277	0.562	0.521

3.9.2 Animal transport

Table 3-24. Transport of animals to and between farms during production. The transport from farm to processing is not included. The capacities of animal trucks is adapted from [420].

	Hauling to the farm			Hauling between farms		
	animal type	[animal / vehicle]	[km]	animal type	[animal / vehicle]	[km]
Dairy				[heifer]	40	200
Chicken (meat)	[chick]	20000	100			
Beef (not on feedlot)				[stocker]	40	100
Beef (on feedlot)	[stocker]	56	100			
Pig	[piglet]	1500	100			
Egg	[pullet]	6000	100			
Turkey	[chick]	6000	100			

3.10 Appendix 3: Estimating fodder consumption

The feed components and associated dry matter (DM) content used to define fodder mixes are shown in **Table 3-25**. The conversion factors used to convert from corn equivalent (CE) feed mass to actual feed mass are shown in **Table 3-26**. Total consumption of concentrate fodder ingredients is shown in

Table 3-27.

USDA estimates the amount of annual grazed biomass in units of CE [43]. This corn-basis mass is converted to actual DM consumed using total digestible nutrients (TDN) for corn and grazed biomass, assumed to be 0.9 and 0.6, respectively (from [46]). Thus, the corn equivalent mass of grazed pasture is multiplied first by 0.87 (the DM content of corn), then by 1.50 (0.9 [TDN / kg DM corn] / 0.6 [TDN / kg DM grazed biomass]); the resulting grazed biomass consumption is shown in **Table 3-28** (1st column).

Annual estimates of harvested roughage fodder production are shown **Table 3-28**. Hay production is reported annually by NASS (of the USDA) [43]. In addition, total harvested roughage production (not including corn or sorghum silage) is reported by USDA in the Census of Agriculture every five years [385], and consumption of total harvested roughage is reported by NASS in CE feed units [43]. However, the annual "all hay" estimate from the annual statistics does not match the AgCensus estimate for the same years. Thus, total harvested roughage production are estimated for this work by multiplying the annual hay estimate from Agricultural Statistics Annual [43] by 1.05 (the average ratio by which the AgCensus estimate of total 'harvested roughage' production is larger than the NASS annual estimate of total hay; years 2002 & 2007 [385, 411]). Annual estimates of hay production (all varieties) are obtained by multiplying the total harvested roughage estimate by 0.89 (the average fraction of all harvested roughage that is hay for years 2002 & 2007 from AgCensus). Estimates of alfalfa hay and non-alfalfa hay were obtained from the total hay estimate by using the alfalfa hay:total hay ratio calculated from Agricultural Statistics Annual [43]. For non-corn and non-sorghum greenchop, alfalfa is assumed to comprise 66% of the total (average of years 2002 & 2007 from AgCensus). Corn silage and sorghum silage production (moist, as harvested basis) are also reported annually by NASS [43].

Table 3-25. Dry matter (DM) content of feeds, mass basis. Concentrate DM from [45, 421, 422]. DM content of hay, haylage, grass silage and greenchop from USDA [385] (Appendix B); DM content for corn and sorghum silages are adapted from various university extension reports [423-426].

Concentrates	**[DM]**
Corn (grain)	0.85
Soybean (whole seed)	0.87
Soybean meal	0.89
Wheat (grain)	0.87
Wheat mill feed	0.89
Gluten feed and meal	0.90
Oats and barley	0.87
Sorghum (grain)	0.87
Cottonseed meal, misc byproducts, linseed meal	0.90
Alfalfa meal, canola meal, fats and oils, peanut meal, rice (millfeed), sunflower meal	0.90
Animal protein feeds	0.95

Roughage	**[DM]**
Hay (“dry”)	0.87
Haylage & grass silage	0.45
Green chop	0.25
Corn silage	0.35
Sorghum silage	0.30

Table 3-26. Corn equivalent (CE) conversion factors (from [383]). Units are [kg corn equivalent DM / kg feed DM]. These are used to estimate mass of individual feed ingredients from the per-animal product CE feed consumption from NASS [427].

(a) Concen-trates	Corn grain	Soy-bean meal	Wheat	Wheat mill feed	Sorg-hum grain	Animal Protein Feeds	Oats	Bar-ley	Gluten feed and meal	Other meal 1[a]	Other meal 2[b]
Poultry	1.00	1.47	1.03	0.76	0.92	1.75	0.88	0.78	na	na	na
Hogs	1.00	1.97	1.01	1.00	0.87	2.37	0.88	0.88	1.64	0.66	1.64
Dairy cattle	1.00	1.67	1.03	0.93	0.97	0.00	0.88	0.98	1.31	0.61	1.50
Beef cattle on feedlot	1.00	1.97	1.03	0.82	0.89	0.00	0.83	0.86	2.11	0.61	2.30
Other beef cattle	1.00	1.82	1.03	0.71	0.97	0.00	0.98	0.98	1.71	0.61	1.41

(b) Rough-ages	Pas-ture	Corn sil-age	Sorg-hum silage	Alf-alfa hay	Non-alfalf a hay	Alfalfa green	Non alfalfa green
Dairy cattle	0.67	0.48	0.47	0.58	0.44	0.49	0.39
Beef cattle on feedlot	0.67	0.72	0.65	0.58	0.46	0.49	0.49
Other beef cattle	0.67	0.60	0.52	0.58	0.49	0.49	0.39

[a] Alfalfa meal, Canola meal, Fats and oils, Peanut meal, Rice (millfeed), Sunflower-seed meal
[b] Cottonseed meal, Misc byproducts, Linseed meal

Table 3-27. Concentrate feeds fed to livestock in US, dry mass (DM) basis. As-fed weight from [214], converted to DM using Table 3-25. The boxes in the top row show aggregation used to report results throughout this work.

	Corn (grain)	Soy-bean meal	Other grains			Corn gluten and wheat bits		Other co-products		Animal bits
	Corn (grain)	Soy-bean meal	Wheat (grain)	Sorg-hum grain	Oats & barley	Wheat mill feed	Gluten feed and meal	Alfalfa meal, Canola meal, Fats and oils, Peanut meal, Rice (millfeed), Sunflowerseed meal	Cottonseed meal, Misc by-products, Linseed meal	Animal-based feeds
1990	98.9	17.8	10.4	8.9	7.8	5.3	0.0	2.4	3.4	2.8
1991	103.0	17.8	5.4	8.1	7.7	5.5	0.5	2.6	3.9	2.8
1992	112.7	18.8	3.4	10.1	6.4	5.6	0.5	2.7	3.4	2.9
1993	100.4	19.6	8.3	9.7	8.5	5.8	0.7	2.9	3.5	3.1
1994	117.2	20.6	6.4	8.3	7.1	5.9	0.1	3.2	4.0	2.9
1995	100.7	20.7	5.3	6.5	6.3	6.0	0.7	3.4	3.8	2.8
1996	113.3	21.2	6.6	11.4	5.6	5.9	1.5	3.2	4.0	3.0
1997	117.7	22.3	7.6	8.1	5.8	5.7	1.6	3.8	3.9	2.7
1998	117.4	23.5	5.5	5.8	5.3	5.7	1.1	3.9	3.5	2.8
1999	121.6	23.5	7.7	6.3	5.4	5.9	1.0	3.9	3.9	2.6
2000	125.4	24.4	5.2	4.9	4.6	5.9	1.2	3.8	3.8	2.3
2001	125.9	25.4	3.0	5.1	3.8	5.6	1.2	3.3	4.2	2.1
2002	119.4	25.0	5.8	3.8	3.9	5.6	1.9	3.2	3.7	2.2
2003	124.4	24.5	3.6	4.0	3.5	5.5	2.0	3.7	3.8	2.5
2004	132.2	26.2	4.2	4.2	3.2	5.5	2.4	3.7	4.4	2.3
2005	132.1	25.9	2.4	3.1	2.8	5.5	2.9	4.0	4.4	2.3
2006	120.1	26.8	4.4	2.5	3.0	5.6	3.8	3.9	4.1	2.4
2007	128.2	25.9	3.6	3.8	2.6	5.7	3.6	4.1	3.7	2.4
2008	114.8	25.4	2.7	5.5	2.7	5.7	3.6	4.1	3.2	2.3

Table 3-28. Supply of harvested roughage in US. DM basis. Derived from annual NASS hay estimates [43] and the every 5-yr AgCensus [385]. See text for description. Hay exports are from USDA [428].

	Grazed biomass	Hay, alf	Hay, non-alf	Green			Hay exports (net)
	Pasture	Alfalfa hay	Non-alfalfa hay	Alfalfa green	Non-alfalfa green	Corn + sorghum silage	Hay exports (all types)
1990	197	60	46	9	5	29	
1991	201	60	51	9	5	27	
1992	200	57	50	9	5	29	
1993	210	58	49	9	5	27	
1994	212	59	51	9	5	30	1.0
1995	214	61	52	9	5	26	1.2
1996	206	57	52	9	5	29	1.2
1997	199	56	54	9	5	32	1.2
1998	199	59	51	9	5	31	1.3
1999	191	60	55	10	5	31	1.5
2000	183	58	53	9	5	33	1.8
2001	184	57	56	9	5	34	1.7
2002	186	51	56	9	5	34	2.2
2003	178	54	60	10	5	35	2.3
2004	179	53	61	10	5	35	2.0
2005	182	53	55	9	5	35	2.1
2006	187	50	52	9	4	35	1.9
2007	189	49	57	9	5	35	1.9
2008	183	50	56	9	5	37	2.0

Table 3-29. Assumed concentrate feed makeup for cattle [mass fraction, DM basis]. It is assumed that cattle only eat vegetable-based feeds.

	Corn (grain)	Soybean meal	Other grains	Corn gluten and wheat bits	Other co-products
1990	0.55	0.10	0.23	0.03	0.09
1991	0.56	0.10	0.20	0.04	0.10
1992	0.60	0.10	0.17	0.04	0.09
1993	0.53	0.10	0.23	0.04	0.10
1994	0.58	0.10	0.18	0.03	0.11
1995	0.55	0.11	0.17	0.04	0.12
1996	0.57	0.11	0.17	0.05	0.10
1997	0.57	0.11	0.16	0.05	0.11
1998	0.59	0.12	0.14	0.05	0.11
1999	0.59	0.11	0.14	0.04	0.11
2000	0.61	0.12	0.12	0.05	0.11
2001	0.62	0.13	0.10	0.05	0.11
2002	0.60	0.13	0.11	0.06	0.11
2003	0.61	0.12	0.09	0.06	0.12
2004	0.61	0.12	0.09	0.06	0.12
2005	0.62	0.12	0.07	0.07	0.12
2006	0.58	0.13	0.08	0.09	0.13
2007	0.60	0.12	0.08	0.08	0.12

3.11 Appendix 4: Overview of animal product LCA studies

Dalgaard et al. perform a consequential LCA of soymeal produced in Argentina and delivered to Rotterdam harbor [372]. The impact of soybeans at the farm gate is 640 g CO_2eq / kg soybean. When calculating the impact of the soymeal specifically, the authors compare the system expansion method with economic value-based allocation. The choice of marginal oil in the system expansion (see **§3.4** for explanation) affected the calculated impact of soymeal: considering palm oil as the product replaced with soy oil results in an impact of 720 g CO_2eq, whereas if rapeseed (canola) oil is considered as the offset oil, the impact of soymeal is 340 kg CO_2eq/ kg soy meal. Since palm oil is associated with lower GHG impact than rapeseed oil, the credit received by the soymeal for displacing palm oil is smaller than if the soy oil co-product instead replaces rapeseed oil.

Posner et al. compare performance of a low-input crop system (basically, organic) with practices that corresponded with widespread conventional practices in Wisconsin, over about 20 years. Conventional corn yielded 8.6 – 9.4 Mg DM/ ha, and the organic system yielded 91% as much. Conventional and low-input soybean yields were 3.6 and 3.3 Mg DM / ha, respectively. An important insight from this work is that yields in the organic system varied more year-to-year, as wet spring weather made mechanical weed control difficult. “Good weed control years” yielded as much as conventional, whereas during the one third of years when weed control was a serious problem, organic yields were 74% of the conventional yields. Grazing land was estimated to yield 11.2 ± 2.2 Mg DM/ ha, while first-year established alfalfa yielded 9.0 Mg DM/ ha in the conventional system and 9.9 Mg DM/ ha in the low-input system [429].

van der Werf et al. report that 1 kg of a weighted average pig feed is associated with 530 g CO_2eq [430]. Pelletier calculates the GHG impact of broiler feed is about 560 g CO_2eq/ kg of poultry feed, with 40% of the impacts coming from poultry-processing co-product feed. These poultry co-product feeds made up 7.5% of the modeled feed mix [398].

Ogino et al. estimate life cycle impact of feeds derived from food residues [227]. Liquid feeds had impact of 270 g CO_2eq/ kg DM feed. The dry feed and incineration systems each had impacts of about 1070 g CO_2eq/ kg DM feed (note that for the incineration scenario, feed is produced from crops, and N_2O is assumed to be ~0.7% of N added).

Fiala [431] estimates future GHG emissions from meat production, using emission factors of 14.8, 1.1 and 3.8 kg CO_2eq / kg beef, chicken and pig, respectively (carcass weight). These emission factors are taken from [93, 432, 433]. Using these factors, Fiala estimates a total annual emission of 1.3 Gt CO2-eq in the year 2000, while noting that an FAO report estimates total emissions of about 7 Gt / yr [301] (see **Figure 3-2**). The author also reports that "Producing 1 kg of beef thus has a similar impact on the environment as 6.2 gallons of gasoline, or driving 160 miles in the average American mid-size car." (p413).

Kumm argues that producing pork organically is less sustainable than conventionally, and that producing meat organically uses more land than conventional; organic meat production should perhaps be limited to use land and byproducts that cannot be used in any other way [434]. This argument is similar to assertions that energy crops should only be produced on lands not suitable for food production.

Westberg et al. present an observation-based inventory of CH_4 from cattle in the US [435]. Clemens reviews options for reducing GHG emissions from animal husbandry [436], and note that anaerobic digestion of manure might offer the most efficient GHG abatement (see **Chapter 5** for analysis).

Phetteplace et al. estimate GHGs from beef and milk production systems in the US [437]. The cow-calf through feedlot system emits 15.5±2.3 kg CO_2eq per kg live weight (LW) gain (8.4, 6.4, and 1.3 kg CO_2eq/ kg LW gain from enteric fermentation for cow-calf, stocker, and feedlot cattle, respectively), with 6.1 kg CO_2eq / kg LW from enteric fermentation. The dairy system emits 1.09±0.2 kg CO2-eq/ kg milk. Total GHG are 20.6, 14.4, and 5.66 for each.

Subak [432] compares GHG emissions of beef production systems. The feedlot-based system emits ~15 kg CO_2eq/ kg of beef, while the pastoralist system emits ~8 kg CO_2eq/ kg beef (kg beef yield / kg live weight is 0.54 and 0.61 for the feedlot and pastoral system, respectively;

calculated from data in Table 2 in [432]). The opportunity loss for carbon storage associated with feed production in the feedlot system accounted for about one third of the impacts reported.

Verge et al. [438] present an inventory of GHG emissions from the Canadian beef industry. In 2001, the GHG emissions associated with producing one kg of LW were 10.4 kg CO_2eq; down from 16.4 in 1981 (mainly because of more feedlot production, which reduces enteric CH_4 emissions).

Stewart et al. [439] estimate the GHG emissions of beef production using a whole-farm point of view. Between 4 and 10 t CO_2eq are emitted per tonne of protein exported from the farm (animal protein contents were assumed to be between 0.18 and 0.2 kg protein / kg LW [p 372]).

Casey and Holden estimate the GHGs associated with three different types of beef production in Ireland [308]: conventional, agri-environmental, and organic beef are associated with 13.0, 12.2, and 11.1 kg CO_2eq/ kg LW, respectively.

Ogino et al. report that one beef calf is associated with 4550 kg CO_2eq [440]. Ogino et al. report that one beef cattle ready for slaughter (raised from a calf) in Japan is associated with about 6,000 kg CO_2eq; using a meat yield of 40%, the impact is 32 kg CO_2eq/ kg beef. This high value is partly due to the long feeding duration (26-28 months) [441].

de Boer [442] reviews the environmental impacts of organic and conventional milk. GWPs range from 0.7 to 1.3 kg CO_2eq/ kg milk. Garnsworthy investigates how fertility rates affect environmental performance of dairy production, and suggest that meeting targets of first calving can reduce methane emissions from dairy herds by 24% in the UK [443].

Thomasson et al. assess milk production. They study the impacts of different production techniques [444] and the differences between the attributional and consequential LCA methods [445]. With the attributional method, the authors allocate co-product impact according to mass and/or economic value. The consequential method is used to estimate the impact associated with a change in production of a functional unit. With this method, system boundaries are expanded to include affected systems (markets). The authors find the consequential method results in a lower impact per functional unit than the attributional method. The avoided burden of not producing some meat (because of the meat byproduct of the milk) is higher than the economic value of that meat. Impacts of 1 kg of milk (energy and protein corrected) from a conventional farm in the Netherlands was 1.6, 1.6, and 0.9 kg CO_2eq for the mass-based attributional LCA, economic-based attributional LCA, and consequential LCA, respectively [445].

Cederberg reports 1100 and 940 kg CO_2eq/ Mg milk for conventional and organic milk production in Sweden [446]. Building and machinery are excluded, as were medicines, washing detergents, and other stable supplies such as disinfectants and salt. "Including machinery would give rise to a factor of uncertainty in the comparison, since conventional milk production to a much greater extent depends on imported concentrate feed." (p 51). A co-product of dairy production is calves, from which milk and meat is obtained. Based on the causal relationship between milk and calf production, 15% of the impact was allocated to calves. Allocation in the feed production system was based on economic value, except in the case of land use, for which allocation was based on mass. Methane contributes about half of the GHG impact for conventional milk production, and more than half for organic. Nitrogen oxide emissions are based on 1.3% and 1.6% emissions factors for conventional and organic, respectively.

Haas et al. compare intensive, extensive and organic grassland milk production in Germany and find GWP impact of between 0.9 and 1.4 kg CO_2eq / kg milk, with the extensified system having lowest impact [447].

Verge et al. calculate that one kg of milk produced in Canada is associated with 1 kg CO_2eq GWP [448] (Note that N_2O emissions from legume forages are discounted [449]).

Eide uses economic allocation for feeds and dairy products, and finds that producing 1000 liters of milk is associated with between 500 and 600 kg CO_2eq [450]. Lovett et al. model GHG emissions from various dairy systems [451]. They report between 0.8 and 1.4 kg CO_2eq / kg milk. Hospido et al. report similar impact for milk production in Spain [452].

Chianese and Rotz review and model GHG emissions from dairies, including C uptake during crop production and C emission via animal respiration [453, 454].

Basset-Mens reports decreased eco-efficiency for concentrate-based dairy production in New Zealand, when compared to low input, pasture passed production ([455]. On the other hand, Capper et al. argue that using the rBST improves milk production efficiency and reduces environmental emissions and water use of dairy production [456].

4 Manure management: Engineering-economic analysis of energy harvesting

Abstract. Concentrated animal feeding operations (CAFOs) can be thought of as animal cities that require waste treatment. Anaerobic digestion (AD) is a manure treatment technique that produces biogas fuel, a mixture of methane (CH_4) and CO_2. Digesting manure reduces odor, making subsequent processing (e.g. separation) more feasible. Nutrients are not removed during the process, so adequate nutrient management is required with or without AD. The cost of biogas production is calculated to be $2-4 per GJ (HHV). Grants covering up to 50% of the capital cost have subsidized installations of AD systems in the US. The electricity-generation portion of AD systems costs about 5¢/ kWh (not including cost of biogas production). The value of electricity produced, including the technical ease of interconnecting with the existing electric grid, is a determinant of AD system economics, but GHG abatement, improved nutrient management, and fiber supply can also provide revenue. Alternatively, biogas may be upgraded to pipeline-grade methane, at a cost of about $4/GJ, not including biogas cost. Nitrogen oxide (NOx) emissions are restricted in some agricultural areas, so NOx could be a concern for AD system developers and regulators. NOx emissions from a biogas engine are likely to be less than 5% of the nitrogen emitted (mostly as ammonia) from manure and fodder on animal farms. Finally, the cost of manure treatment with AD is compared with the overall cost of producing animal products. An AD manure treatment system in which biogas is flared without energy recovery adds 0.5%-1.5% to the cost of production for dairy- and pork-producing operations.

4.1 Introduction

Manure is a byproduct of animal farming that can be an asset or a liability. It can pollute the environment with nutrients and pathogens and annoy neighbors with its odors. It may also provide value via electricity, heat, fertilizer and fiber. Keeping animals comfortable and healthy promotes growth and production, and so managers of animal operations have incentive to manage manure for animal comfort. As with many industries, the degree to which costs of manure management can be externalized by animal producers influences the benefit-cost ratio for manure management alternatives.

Liquid manure management is common on many concentrated animal feeding operations (CAFOs), especially at pork and dairy operations, and some egg farms. As farm size increased, large-scale liquid and slurry manure storage facilities were promoted to avoid spreading manure on fields during sensitive seasons and weather episodes. The current motivation to support manure treatment via anaerobic digestion is multifaceted, since digesters can: reduce GHG emissions, make animal farms less of a nuisance for neighbors, and aid manure nutrient management. Not least, a few spectacular failures of manure storage lagoons (some during periods of heavy rain) provided a catalyst for government regulation of manure management.

This chapter provides an analysis of manure production, energy capture and regulatory issues, along with an economic framework for assessing manure to power (MtoP) projects. First, the magnitude of manure production in the United States (US) is reviewed. Then, an economic-engineering analysis of energy harvesting from manure is presented. Finally, the costs and revenues of manure management with MtoP are compared to animal product revenues on representative animal farms.

4.2 Animal manure in the US

Manure contains nutrients, organic matter and energy, which are all potentially valuable (**Table 4-1**). Manure nutrients essential for plant growth have long been recycled as fertilizer for crops and pasture. However, animal production is now sufficiently concentrated in many areas of the country that manure nutrients cannot be safely applied to available cropland [289, 457, 458].

Dairy and hog production has become more concentrated over the last 20 years, with the result that a higher fraction of total production occurs on larger farms (**Figure 4-1**).

Table 4-1. Representative value of manure resources: nutrients, energy and fiber. Value is shown per tonne of manure dry matter excreted (DMx). Ranges reflect differences in manure nutrient and energy content for different animals, and the value of the animal products being produced. Nutrient value based on 50% of N, P and K available as fertilizer, with average fertilizer prices from years 2001-2008. See Appendix for manure production, resource values, and animal product values (§4.8.1 and §4.8.2).

Resource	Value per tonne manure DMx	Value per $100 of animal product
Nutrients (N,P,K)	$28 - $54	$1.5 - $3.2
Energy	$21 - $51	$1.5 - $3.2
Fiber	$ 4 - $ 5	$0.2 - $0.6

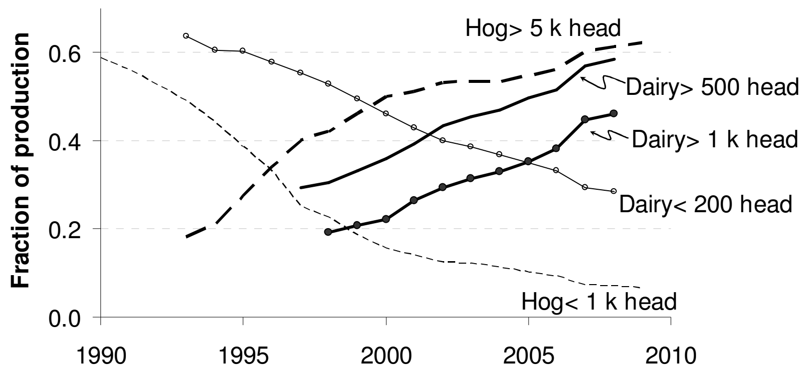

Figure 4-1. Change in hog (two dashed lines) and dairy (three solid lines) production in US on smaller and larger farms. More than 60% of hog meat is currently produced on farms with more than 5,000 animals, and over half of milk is produced on farms with more than 500 cows [43].

Animals in the US excrete[5] more potential manure pollutants (or resources out of place) than humans (**Figure 4-2**). If all manure from dairy cattle, chickens, beef cattle on feedlots, pigs and turkeys were digested, about 14 GWth of biogas could be produced (thermal energy basis, HHV), or 0.5 EJ of thermal energy per year. This estimate is lower than the 1 EJ reported by Cuéllar and Webber [459], who include all cattle, including those which are not on feedlots. The energy contained in manure from the cattle not on feedlots (mostly beef cows and calves) is about 0.4 EJ / yr, but the manure from these animals is not generally available for use as an energy feedstock (**Table 4-2**).

[5] 'Void' is a more precise term, since feces include undigested matter that was never absorbed (and thus excreted) by the body. However, 'excrete' is more widely used as a synonym for defecation and urination, and is adopted here.

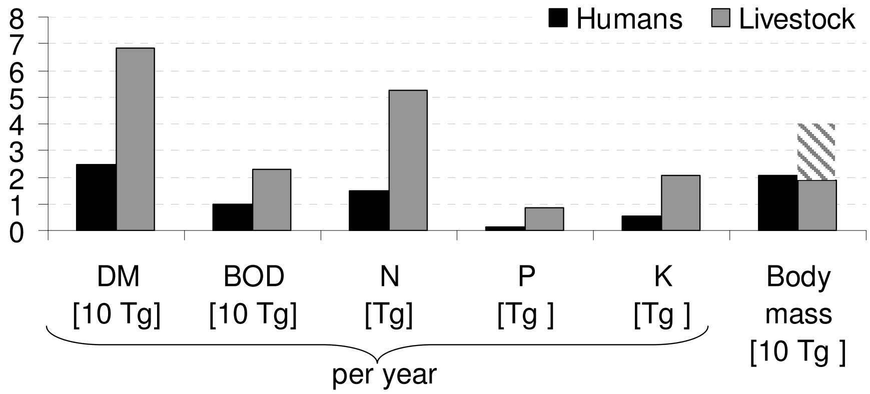

Figure 4-2. Annual production of animal and human waste in the US, yr2007. Manure production from Appendix (§4.8.2), animal populations from Chapter 3, human population from US Census [47]. "Animals" includes dairy cattle, beef cattle on feedlots, chickens (broilers and layers), pigs and turkeys. DM=dry matter; BOD = biological oxygen demand. The cross-hatch area for animal "Body mass" represents beef cows and calves (not included in other categories).

Table 4-2. Energy content and biogas production potential of animal manure in US (GWth = gigawatt, thermal energy basis, HHV). The energy yield from biogas is 30% - 60% of the total energy content of manure dry matter (when combusted). Calculated from manure properties in Appendix (Table 4-6) and animal populations and production in Chapter 3.

		Dairy cattle	Chickens (meat)	Beef cattle on feedlot	Pigs	Turkeys & egg	sub TOTAL (left)	Other beef cattle (not on feed)
E contained in solid matter	[EJ / yr]	0.55	0.17	0.15	0.12	0.06	1.06	1.21
	[GWth]	17.5	5.5	4.7	3.8	2.0	33	38
Biogas E	[EJ / yr]	0.18	0.11	0.06	0.07	0.04	0.45	0.37
	[GWth]	5.6	3.4	1.7	2.4	1.3	14	12

Manure management techniques have a large impact on the environmental discharges from animal farms. Practices and technologies are diverse, and will continue to develop and change. Collecting and managing excrement to improve environmental conditions is not new. In urban environments, human sewage is managed by connecting toilets to treatment facilities via pipes. The magnitude of manure produced on a CAFO is comparable to sewage production from a moderate size human town (Appendix, **§4.8.2**, **Table 4-8**). Large farms can be thought of as animal cities, and cities need waste treatment facilities.

Energy can be harnessed from manure with methane collection via anaerobic digestion, or by other thermochemical processes (including combustion or gasification). Electricity production is often the focus of MtoP schemes, but the energy can also be used for space and water heating, or to supply heat for industrial processes (e.g. ethanol production). Organic matter in manure (fiber) also has value and can be used as bedding material. When spread on fields, manure increases organic matter content, microbial diversity, and improves physical characteristics of soil [460].

4.2.1 Manure management strategies

At dairy operations, cows are often kept in free-stall barns or feedlots, where the animals can walk around the barn or lot and lie in bedded stalls. Mechanical scrapers may be used to collect manure from walking lanes (**Figure 4-3**); vacuum trucks may also be used. After some dilution with water used to clean the milking parlor, the manure stream is about 10% or 11% solids by weight. Other dairy farms use water to flush manure from the barns or alleys; the manure slurry on such operations can be as dilute as 1% solids. With either system, the slurry often ends up in storage (lagoons or large tanks) and is applied to the land or disposed of when appropriate.

Cattle feedlots in relatively dry climates often manage manure as a solid. Manure is allowed to accumulate in a dry lot (parts of which may be covered to protect animals from sun, rain and/or snow) and is periodically removed.

On many hog farms, pigs are housed in barns with slatted floors that allow excrement to fall into pits below. Farms that utilize “pull-plug” manure management flush manure from pits below the hogs. The slurry, which is about 3% to 6% solids, then travels to a storage lagoon. With “deep pit” manure storage, the pits below the hogs are emptied every few months and the manure is applied directly to the land—a process that can be very smelly. Conveyers to remove manure from housing daily may also be installed below slatted floors [461]. Such systems may allow some liquid to drain from solids before it leaves the barn.

Most poultry farms handle manure as a solid or semisolid litter. If birds are caged (as most egg hens are), the cages are usually stacked, or are elevated, so the litter falls below the cages. The manure may be allowed to collect in pits and piles, or it falls onto conveyer belts for continuous removal. Depending on the handling method and amount of time manure is allowed to collect, the moisture content of manure from caged birds is usually less than 60%. Some hen barns are flushed with water, producing a more dilute manure stream (>90% moisture) requiring tank or lagoon storage.

If the birds are housed in an open barn (as with broiler chickens, turkeys, and some "free range" layers), manure collects in a pack on the floor. This litter pack, which includes bedding material such as wood shavings, dries as it accumulates, so moisture content is often lower than 50% when the litter is removed.

Figure 4-3. Cattle housed in free-stall barn with mechanical scraper in alley. Note the stalls are slightly shorter than a cow lying down, so even manure from an animal in a stall is deposited mostly in the alley. (Image: Kyle Meisterling)

4.2.2 Manure regulations and environmental discharges

The Clean Water Act protects the nation's surface waters by requiring point sources of pollution to have National Pollutant Discharge Elimination System (NPDES) permits. Under the Clean Water Act, CAFOs[6] are identified as point sources, and so CAFOs which discharge are regulated, and required to obtain a permit to discharge [462]. The US Environmental Protection (EPA) provides that a "CAFO proposes to discharge if it is designed, constructed, operated, or maintained such that a discharge will occur", and the EPA "contemplates" that CAFO operators will perform an objective assessment of whether discharges will occur [463].

The original CAFO regulations promulgated in yr2003 required all CAFOs to obtain permits [464], under the assumption that all CAFOs had at least the potential to discharge. A subsequent court decision in yr2005, however, found that the EPA does not have the statutory authority to require all CAFO operators to apply for a discharge permit [465]. The decision also requires that operators proposing to discharge submit nutrient management plans. The EPA authorizes individual states to issue permits to CAFOs, and if a state is not authorized, the US EPA is responsible for issuing permits.

Major air emissions from animal facilities include volatile organic compounds, ammonia, methane, hydrogen sulfide and particulate matter. These emissions are often associated with animal excrement, but feed has also been indicated as a major source of ozone precursors from dairies in the central valley of California [466]. The relatively high moisture content of dairy

[6] A concentrated feeding operation (CAFO) is roughly defined as an animal operation with over 1,000 Animal Units (AU). An animal unit corresponds to 1,000 pounds of live animal weight – about the weight of a mature beef cow. Animal feeding operations (AFO) are defined by the EPA as "agricultural enterprises where animals are kept and raised in confined situations." CAFOs are a subset of AFOs and are regulated by the EPA.

rations including silage and green chop is associated with higher ozone precursor emissions (e.g. ethanol and aldehydes) than drier grain and byproduct feeds. Odor is also a major concern.

There are three legislative acts pertaining to air emissions which may be applicable to animal operations: the Clean Air Act (CAA), the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), and the Emergency Planning and Community Right-to-Know Act (EPCRA). The CAA requires emitters of sufficient magnitude to have permits; CERCLA and EPCRA require the person in charge of a facility to report emissions. The nature of emissions from animal farms makes it difficult and expensive to determine exactly how much of a pollutant is being emitted. For this reason, the application of CAA emission limits and CERCLA and EPCRA reporting requirements has been limited, and regulatory action will likely involve approved technologies and practices, as opposed to continuous monitoring with emission fees.

In order to better understand emissions sources, the US EPA and over 2000 AFO operators entered into an agreement whereby the operators allow air emission monitoring, pay civil penalties (if limits are exceeded), and agree to install BACT ("best available control technology") or achieve LAER ("lowest achievable emissions reduction") [467]. Beef open feedlot operators did not enter the agreement, and emissions from manure spreading are not considered. The emissions from animal manure on operations smaller than the CAFO threshold are exempt from reporting requirements of CERLA [468].

The most stringent emission limit for animal operations under the Clean Air Act is likely the 100 pounds per day of ammonia limit. If 100 pounds of ammonia are emitted on any one day of the year, the regulation goes into effect. Using an ammonia estimation tool from the University of Nebraska (http://cnmp.unl.edu/consentagreement.html), it has been estimated that ammonia emissions from buildings housing 200 dairy cows, 2,000 finishing hogs, or 70,000 broilers may exceed the 100 pounds per day limit at least once each year [469]. There are, of course wide variations in emissions due to different farming practices, regional climates, and weather; pH is an important determinant of N emission as ammonia [470, 471]. Thus, pH manipulation and water content management could be key practices in controlling manure NH_3 emissions.

4.2.3 Manure transport

Since manure on CAFOs is produced in a central location, it must be moved from animal housing and storage facilities if it is to be applied to the land. Transport of liquid and slurry manure may occur via hoses or tractor-pulled slurry tanks to be applied on fields near the CAFO.

As discussed above, manure is sometimes a stranded resource, produced in excess of the capacity of the surrounding land. Transport to a location where nutrients can be used adds value. However, transporting manure may involve moving a lot of water, particularly if the manure has been diluted (manure excreted by mammals is 85-90% water; poultry manure is dryer as excreted – about 75% water [415]). If, on the other hand, the manure has had an opportunity to dry, either during accumulation (in a barn or dry location) or during storage, it may be more easily transported by truck than liquid manure. The economics of transporting manure depend on both the value of manure at its ultimate point of use, and on the cost of alternative disposal.

Cooperative animal manure processing is less common, but may be feasible in some circumstances. For example, piping slurried animal manure from animal housing facilities to a centralized digester could be an attractive option in areas with very high densities of animals [472, 473]. Alternatively, manure can be trucked (either in a solid form or liquid) away from the farm for centralized treatment. The nutrients present in the manure may return to farms for use as crop fertilizer, or they may be sent away permanently, to be used where the nutrients are desired, or can be disposed of.

The physical and chemical properties of manure differ depending on diet, species, climate, weather, and management style, so precisely estimating costs associated with manure transport is difficult. In addition, fluctuating fuel prices affect the cost of transport. Poultry litter, which is relatively dry, is routinely transported via truck from CAFOs to land which can be permitted to receive the manure. Liquid and slurry manure transport distances are more restricted, owing to the higher water mass per nutrient.

A range of manure transport costs are shown in **Figure 4-4**. The cost of transporting milk is included as an estimate of the minimum cost for liquid manure transport – milk is a commodity with a high moisture content (about 90%) that is routinely transported considerable distances. While milk transport is distinguished from manure transport by the more homogenous nature of milk and longer shipping distances, milk shipping is an example of a well-developed liquid transport system.

A transport cost of \$5/ tonne of manure (10 – 30 km distance from **Figure 4-4**) for manure with 88% moisture corresponds to a cost per tonne of manure dry matter (DM) of \$42. This cost is roughly equivalent to the value of nutrients in the manure presented previously in **Table 4-1**, illustrating the limitation of cost-effective slurry transportation.

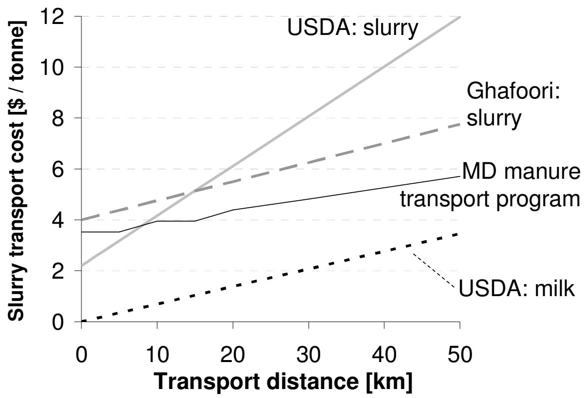

Figure 4-4. Estimates of manure transport costs (yr2005 $). Sources as follows: USDA: slurry [474], Ghafoori: slurry [473]; MD manure transport program [475], USDA: milk [476] (based on average Mideast mileage rate of 5.3 (¢/hundred weight/mile).

4.3 Overview of energy harvesting technologies

In order to make power with manure, it must be collected. The feasibility and cost of an MtoP system is dependent on the manure management practices at a given farm, as well as the landscape of energy infrastructure and demand in the immediate area. For example: What price does the farm pay for electricity? Is the electricity grid able to accept power generated on the farm? Is there a natural gas pipeline nearby that could accept upgraded bio-methane? Is there a commercial or industrial facility nearby that could use the biogas (e.g. and ethanol plant)?

Two technologies are proven to be technically and (sometimes) economically feasible for harvesting energy from manure: anaerobic digestion and combustion. Typically, anaerobic digestion is used to treat wet manure (liquid or slurry) and produce methane as a fuel, although lower-moisture digestion (anaerobic composting) is also used. Combustion is implemented using dry manure (e.g. poultry litter) as a fuel. Other technologies have been tested and show some promise (in particular, gasification via pyrolysis could enable decentralized power generation from solid manure), although they have not, as of yet, proven to be commercially viable for harnessing energy from manure.

4.3.1 Anaerobic digestion (biogas)

Anaerobic digestion (AD) is the process by which bacteria break down and consume organic matter, and produce biogas (a mix of methane and carbon dioxide gases) as a byproduct of their anaerobic respiration, along with ‘stabilized’ organic matter [477, 478]. AD may provide various services, including energy via biogas, manure solids mass reduction, odor mitigation and sanitation. Biogas production occurs naturally in the digestive tract of ruminant animals and in bogs and sediments where moist, organic-matter rich environments lack oxygen. AD for energy recovery utilizes similar microbial processes, and takes place in a sealed container. It is useful to think of a digester as a living organism which can be under- or over-fed, resulting in system failure. Manure is collected from animal confinement areas and pumped to the digester vessel, which may consist of a covered manure lagoon, cement trough, or steel or cement tank(s).

The biogas is typically collected within the digester headspace (space between the surface of the manure slurry and the digester cover), although it may be stored in a separate inflatable bag or in a covered post-digestion manure-storage tank. The methane is often used as fuel on-site, either in a boiler or an engine powering a generator. Alternatively, biogas can be upgraded to pipeline-grade natural gas, and injected into the interstate pipeline for sale and use at a distant location. If biogas is not valuable as fuel, it can simply be flared to convert methane to carbon dioxide. The digested manure has lower odor and improved fertilizer properties compared to raw (undigested) manure, and is typically spread on fields as fertilizer.

A centralized digester uses a similar process, but has an unloading or reception area for the manure and/or other organic wastes being processed. Typically, centralized digesters have additional pre- and post processing vessels to help maximize the production of biogas.

As of yr2009, there were about 125 manure biogas systems operating in the US (most between 100 kW and 1 MW of electrical output) [479]. Many of these have been built with grants covering up to 25% of up-front costs [480]. Many more have been built in Europe, with over 2000 digester systems in Germany subsidized via the electricity produced [481]. In the US, the concrete trough plug-flow digester design commercialized by the company RCM Digesters (now RCM International LLC) has been instrumental in demonstrating that AD managed at the farm scale is feasible in the US. The relatively simple plug-flow design has provided reliable operation, mostly on dairy farms, when properly constructed. Digesters with more advanced processing technologies, monitoring, and gas clean-up may provide more operational flexibility. There are now many digester developers in the US, with some licensing European technology. The economics of biogas are investigated in **§4.4**.

4.3.2 Combustion

Animal manure that is sufficiently dry can be incinerated to produce steam and power a steam-turbine generator. Poultry litter is a good candidate for direct combustion because of its low moisture content. Most broiler chickens and turkeys are raised in poultry sheds, where manure accumulates along with bedding such as wood shavings, in a pack on the floor. Broiler litter has moisture content between 15 and 30%, ash content between 10 and 30%, and a higher heating value (HHV) around 10.5 MJ/ kg (HHV of bituminous coal is 28 MJ / kg) [482-486]. The litter pack is collected periodically from the poultry sheds and trucked to the generating plant.

There are four poultry litter combustion plants in the United Kingdom (although one now burns meat and bone meal instead of litter), one plant in Minnesota, and various other projects under development [487]. These biomass plants (10-65 MW electric output) need a substantial, dedicated supply of biomass. Thus, fuel supply stability is a particular concern for combustion project development. Capital cost for the plants is between \$3,000 and \$4,000 per kW of electrical capacity [488].

Although beef cattle feedlots could be a large source of dry manure (**Table 4-2** above), feedlot manure as fuel for combustion is less than ideal. On feedlots, manure is often mixed with dirt, so ash levels of collected manure can reach 50%. However, stabilizing feedlot surfaces (e.g. with fly ash from coal-fired power plants [489]) could make energy production more feasible by improving fuel properties of collected manure.

In cases where there is limited land locally on which to spread manure, incineration or gasification could provide a cost-effective waste management solution. Phosphorous present in the original litter is concentrated in ash and can be transported at a much lower cost than the original material.

4.3.3 Fuels from thermochemical conversion

During pyrolysis, gasification and liquefaction, the atoms and molecules that make up organic matter (biomass) are rearranged in order to produce useful fuels and/or raw materials. Pyrolysis occurs without atmospheric oxygen and is driven by elevated temperature. During gasification, some oxygen is added to produce syngas (carbon monoxide and hydrogen gas). These processes are referred to as thermochemical conversion[7] (TCC). The products from TCC include gaseous fuels (as in the case of pyrolysis and gasification), solid char (as with pyrolysis) and liquid fuel (as with fast pyrolysis and liquefaction) [490].

Syngas can be utilized in a reciprocating engine to produce electricity [491], and the char remaining from pyrolysis can be used as a soil amendment which improves nutrient retention and sequesters C [492]. Liquid fuel produced on-site with fast pyrolysis can be transported to a central facility for further processing and/or utilization [493].

In general, these conversion technologies have not yet been proven effective for commercial energy harvesting from manure. Poor feedstock properties (i.e. high ash content of dry manure, high alkali content of ash, dilute liquid manure, and inconsistent feedstock moisture content which makes handling difficult) and fuel characteristics (i.e. "tars" in producer gas, high water content of liquid fuel from fast pyrolysis) currently make TCC processes unfeasible for many animal operations. While each of these conversion technologies is being applied in commercial settings with other biomass (mostly wood), they have not been proven with manure. If CAFO management practices can produce manure with better, more consistent fuel properties, these TCC processes may provide value.

4.4 Anaerobic digestion: engineering-economic analysis

In practice, a digester is a waste management technology that offers the potential for various revenue streams. An anaerobic digester can be justified on economic grounds if it provides an acceptable rate of return on investment, or if it provides a least-cost solution to environmental

[7] Combustion is also a TCC process, but does not produce a fuel (as with pyrolysis, gasification and liquefaction), and so is discussed separately.

pollution problems. Thus, the attractiveness of an investment in a digester will depend on the point of view of the decision-maker. For example, a farm owner may be willing to pay a fee for manure treatment, but may not want to involve herself with the planning, interconnection, capital cost, and maintenance requirements of a digester system. In this case, the fee that the farm owner is willing to pay may be a revenue source for an independent digester developer.

An important driver for manure AD development has been government policy (both at the state and federal level), motivated by the nuisance and risk of storing large quantities of manure. The ability to pump liquid manure has been an integral component of large dairy and hog operations. Biogas systems are compatible in this paradigm of liquid manure management, thus AD has been subsidized. AD systems are not the only option for managing manure, but benefits such as locally produced energy from waste, and reduced impact from large animal farms, have attracted political support.

The imperative to improve manure management is also a result of the potential for large manure storage lagoons and subsequent field spreading to impose a nuisance on neighbors. For example, juries have awarded nuisance settlements to neighbors of exceptionally odorous manure facilities. The imperative could also be related to manure management permitting - CAFO regulations that impose fines on Clean Water Act violators might cause operators to carry out an assessment of the chance of pollutant discharges. An AD facility may thus provide benefits to a farm operator that are difficult to monetize - namely, the reduced likelihood of costly settlements, and access to operating permits.

Manure, the main substrate in farm digesters, is not a particularly high CH_4-yielding digester feedstock, in terms of CH_4 production per kg of dry matter added to the digester. However, other organic substrates can be co-digested, including corn silage, crop byproducts, and food processing wastes such as cheese whey, vegetables and waste fats and oils. These non-manure feedstocks often have substantially higher methane yield per mass, but their availability varies from region to region. In addition, they bring additional nutrients which must be utilized or managed.

4.4.1 Operational setup & biogas utilization options

In the US, the majority of AD projects have been designed and built as on-farm systems to produce electricity using biogas as fuel. Since virtually all farms use electricity and are connected to the electrical grid, producing electricity by burning biogas in an engine-generator is a common way to produce revenue from biogas. Electricity may be used on-site, with excess electricity exported to the distribution utility. Alternatively, all the power can be sent to the (sub)transmission system if there is a substation nearby. In this situation, the farm purchases power from the distribution utility just as it would without a digester.

To produce electricity, biogas is most often utilized in a combined heat and power (CHP) reciprocating engine. Microturbines and Stirling engines have been experimented with, but they are more expensive than reciprocating engines. Some heat from the CHP unit is used to keep the digester at the proper temperature, and may also be used to pasteurize feedstock from off-farm, and provide some heat for the farm if needed. Biogas-powered electricity generation equipment is well-developed in the US and abroad. It is not, however, the only way market biogas energy.

Another option to capture value from an AD project is to produce pipeline-grade methane by removing CO_2 (and other low-concentration impurities) from the biogas. The methane can be injected directly into the natural gas grid if location allows (see map of natural gas pipelines in **Appendix, §4.8.3**). Building a pipeline from a farm digester to a natural gas pipeline is another option, as is trucking compressed methane. However, these latter two options involve considerable additional costs.

A key issue for digester developers and utilities is whether a distributed electricity generator improves grid function or whether it imposes a cost. Producing electricity from a waste is politically easy to promote, and electricity is relatively fungible, so the idea of connecting a MtoP system to an existing power system is conceptually attractive. Expensive feasibility studies required by utilities to determine the effect on the grid can act as barriers to distributed generation development [494]. On the other hand, distributed generation located on stressed radial feeder power lines may improve grid operations. In many locations, the grid was designed for one-way power flows, from the utility to the customer, so some MtoP installations could require distribution system upgrades [495, 496]. Furthermore, electricity prices vary across the US (**Figure 4-11** in **Appendix**). The interconnection of distributed electricity generators is being standardized in IEEE series 1547 [497]. Determining required distribution line upgrades must be determined on a project-by-project basis (e.g. [498]), but standards, along with accumulating experience, should reduce the costs of the necessary information-gathering activities [499]. It is clear that the benefits and costs of producing distributed power are location-specific.

In addition to site-specific electricity and equipment costs, the cost of an AD system depends on:

- The nature of the material being digested (i.e. whether it is a dilute liquid, slurry, or semi-solid material, along with the methane yield of the material and presence of inhibitory materials, e.g. antibiotics)
- Whether the material must be transported and/or processed (sorted, chopped, shredded, slurried, etc)
- The digester technology being used (anaerobic lagoon, complete mix digester, high-solid digester) and temperature at which the digester is operated
- Final use of the gas (e.g. used as fuel on-site in a reciprocating engine; upgraded to pipeline grade natural gas; flared).

4.4.2 Digestate management

Digestate is the manure that remains after digestion. During AD, between 25% and 50% of manure dry matter (DM) is converted to biogas (lower for ruminants than for poultry and pigs), while manure nutrients (N,P,K, etc) are retained in the digested manure. Furthermore, since manure as excreted by animals is mostly water, the total mass reduction associated with AD is usually less than 10%. Thus, AD does not make manure disappear, and even with AD, manure still has to be utilized or disposed of.

A digester system can be designed with a manure storage structure that is covered and also serves as a secondary digester and gas holding tank. The liquid or slurry is applied to fields when appropriate. The fiber present in manure can be separated from the slurry and reused. It is common for dairies to use the separated solids for bedding, potentially offsetting purchases.

4.4.3 Equipment and process costs

Because electricity has been a focus of AD system development, total system capital costs are often reported in terms of electrical generation capacity. **Figure 4-5** shows the installed capital costs of agricultural, farm-based anaerobic digester systems built in the US through 2004, and in Germany. Most of these digesters are mesophilic (operate at around 100° F), have capacities between 70 and 500 kW electric (kWe) and cost between $3,500 and $4,500 per kWe to construct and start-up. As would be expected, the cost on a per-kWe basis is quite large for small digesters. The per-kWe cost decreases up to about 100 kWe, beyond which costs are scattered around $4,000/ kWe. The costs shown in **Figure 4-5**, including the lack of economies of scale above 100 kWe, are consistent with more recent costs reported by AgStar, an EPA agency that promotes digester installation for methane emissions abatement [500-502].

In terms of component costs, biogas production and utilization are two distinct activities. The digester itself (including pumps, tanks, pipes, engineering and startup) typically makes up over half of the system cost. Some manure handling equipment is necessary on animal farms with or without the digester, and so this equipment (which is often in-place if an AD system is added to an existing farm) is not usually included with the capital cost of an AD system. The cogeneration equipment makes up about a third of the costs, with other costs such as buildings, energy distribution on-farm and engineering making up the balance (**Figure 4-6**).

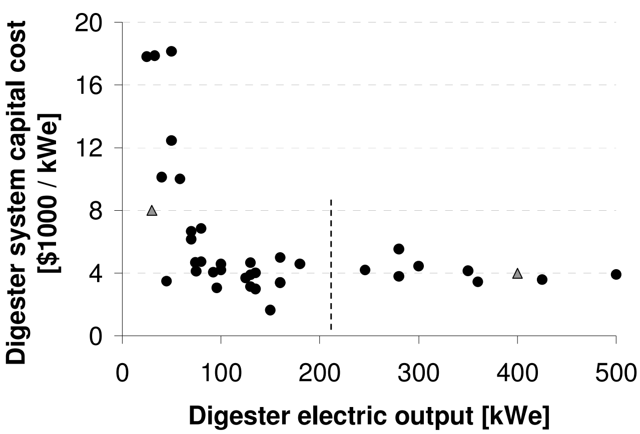

Figure 4-5. Capital costs of anaerobic digesters in US through 2004 (dots) and Germany (triangles). Includes only equipment necessary for digestion and electricity production (does not include solid separation or composting equipment) [503-514]. Notice that costs are scattered around $4,000 per kW electrical capacity (kWe) for digesters sized around 100 kWe and larger. The vertical dashed line indicates the approximate electrical output from manure on a dairy farm with 1000 milking cows and 450 replacement heiffers. Costs adjusted to yr2005 $ with CPI [76].

Figure 4-6. Cost breakdown for on-farm digester systems. 'Digester' includes all equipment necessary to produce and flare biogas. 'Other' includes buildings and any gas or electricity distribution necessary on-farm. This breakdown is representative of a digester sized between 100 kW and 1 MW. Standard deviation of 8 data sources shown [506, 511-517].

Since digesters are often installed for waste treatment, assigning the total cost of the system to the energy produced is generally not a fair assessment of the cost of the recovered energy. Thus, the costs of installing and operating the equipment necessary for biogas use or upgrading are presented separately from costs of the digester itself.

Like any other waste treatment facility, periodic expenditures for monitoring and maintenance are required to keep a digester system running properly. Because farming practices and manure properties differ, and since there are not many systems that have operated for over 10 years, O&M costs in practice are not well quantified, and vary by digester design. For example, foreign materials such as sand enter the digester mixed with manure. Some digesters have a self cleaning cycle that results in solids and debris being ejected from within the digester during processing; others require shut down to allow men and equipment to enter the digester to remove the unwanted materials. Likewise, maintenance required for electricity generation equipment will depend on the quality of the gas.

To estimate the cost of producing biogas, I will use the cost breakdowns from **Figure 4-6**, with costs in the "other" category divided evenly between biogas production and electricity generation equipment. It is assumed that the fixed operating costs for the digester amount to 5% of the non-electricity generation capital cost annually. Capacity factor for the digester is 90%, and all capital expenditures not paid for by grants are financed at 7% over 15 years. Using these parameters, the cost of biogas from AD is $3.3/ GJ (HHV). This cost includes only the equipment necessary to produce biogas, and does not include the cost of electricity generation or methane purification equipment.

I assume that variable operating cost related to electricity production is 2 ¢/ kWh (which accounts for the necessary engine maintenance and rebuilds and operation of gas conditioning equipment), the capacity factor of the generation equipment is 85%, and that the electrical generation efficiency of reciprocating engines burning biogas is 35% (HHV basis). Thus, the cost of converting biogas to electricity, not including the cost of the biogas, is about 5 ¢/ kWh. **Figure 4-7** shows the sensitivity of electricity conversion cost to changes in the capital cost of the generation equipment (most often a piston engine driving a generator).

For a distributed electricity generator, tying into the (sub)transmission system, as opposed to distribution lines, could streamline interconnection. Under such circumstances, a power line might be constructed to carry the electricity from the digester to a transmission substation. In yr2000, a 69 kV single circuit line cost about $300,000 per mile to construct [518]. Extending an interconnection line to a transmission substation would probably be economical only if interconnection to the distribution system requires expensive upgrades, for which the digester developer is responsible, or if the substation is nearby.

When methane is produced from biogas (by removing CO_2), it may then be marketed to the gas grid locally or via pipelines, or can be used as vehicle fuel. **Figure 4-8** shows costs associated with removing CO_2 from biogas. In the figure, the smallest facility shown uses about as much biogas as a dairy with 1,000 milking cows would produce (0.5 MW thermal, or 50 m^3 of biogas per hour). The upgrading cost for small systems can be over \$10 per GJ (HHV). A pipeline built to market methane from a 3 MWth biogas plant would add about \$0.5/ GJ per mile of pipeline (assuming 350 k\$ per mile of pipeline [519, 520], and financing at 10% over 15 years).

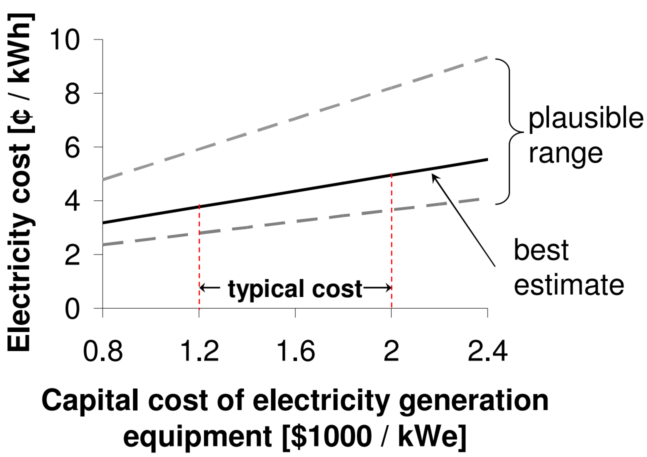

Figure 4-7. Cost of the electricity generating portion of a digester system. Most reasonable estimate assumes: 2 ¢/ kWh variable O&M, 85% capacity factor, 15 year payback, and 7% interest rate on capital. This figure does not include the cost of producing the biogas, but only of using it as fuel for power generation.

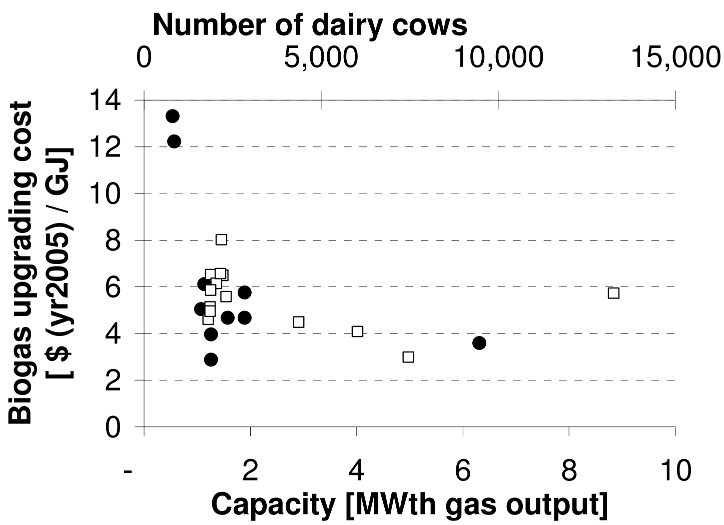

Figure 4-8. Costs of removing CO_2 from biogas in Europe (does not include cost to produce biogas). The cost of gas upgrading decreases as the size of the plant increases. The size of the plant is shown both in MW thermal (MWth) gas output, and as the number of cows from which manure is used to produce biogas. Dots from [521]; boxes from [522].

4.4.4 Revenues and benefits

The value of electricity has been an important – if sometimes contentious – factor in the development of anaerobic digesters. As mentioned above, the need to improve management of large quantities of liquid manure has likely been the most important driver for early AD adoption in the US, but the energy harvesting part of the system has been the most visible outcome of such projects. Concerns about climate change have also driven AD development, whereby methane capture and control yields relatively inexpensive GHG abatement (see **Chapter 5**).

Many biogas facilities have been subsidized. These subsidies aim to: reduce odor and improve manure management on farms (e.g. grants from the Environmental Quality Incentives Program [EQIP] program of the Natural Resources Conservation Service); reduce methane emissions from manure storages (AgStar of the EPA & USDA, and GHG offset markets); and produce low-GHG energy (the Renewable Electricity Production Tax Credit [PTC] and Clean Renewable Energy Bonds [CREBs] for municipalities and cooperative electric utilities). A digester with biogas flare can provide the first two of these three aims, while biogas utilization for energy services provides all three.

While around one third of the energy contained in biogas is converted to electricity, 50-60% of the biogas energy is converted to useful heat. In cogeneration applications, this energy is used for space or process heat [523]. For on-farm applications, this heat may displace some propane or electricity purchases (e.g. hot water for cleaning in dairies and heat for pig and chicken nurseries).

The comparative economics of producing biogas and converting it to either electricity or pipeline-grade CH_4 are shown in **Figure 4-9**. AD systems operate under various ownership and operational arrangements, energy price regimes, and local markets. Thus, the figure is intended to offer insight into the range of cost and important determinants of revenues associated with AD systems.

In **Figure 4-9 (a)**, the annual costs of operating and owning a digester producing 1.4 MWth of biogas are shown, along with potential revenues from selling the biogas as fuel, GHG credits (if the digester reduces CH_4 emissions from manure storage), and fertilizer (if digestion improves fertilization properties of manure, and subsequent application offsets some fertilizer purchase). These three revenue streams are highly conditional.

The methane abatement values in **Figure 4-9** reflect a situation where the farm without a digester would emit CH_4 from manure storage at a rate equal to half of the digester methane production –

thus, digester installation is responsible for CH_4 abatement, and receives emissions credits. Emission credits are assumed to have a lifespan of 10 years. In such a circumstance, GHG credits priced at $30/ tonne CO_2 would pay for digester construction and operation (see **Chapter 5** for analysis of GHG impacts of AD).

Digestion can improve the fertilization qualities of manure through mineralization, and the nutrient revenues shown in the figure assume digestion is responsible for fertilizer purchase offsets corresponding to 25%, 10% and 10% of the manure nutrient content of N, P and K, respectively. Digestion may also allow utilization of manure solids as fiber, with an assumed yield of 0.05 t fiber per GJ of biogas produced (calculated from **Table 4-6** in Appendix). The low revenue case for "nutrient & fiber" only considers N; the mid case considers N, P, K priced at averages over yr2001-2008 and fiber priced at $20/ tonne; the high revenue case considers nutrient prices at one standard deviation over the average fertilizer price (see **Table 4-7** in Appendix) and fiber priced at $30/ tonne.

Figure 4-9 (b) compares the annual costs and revenues of using biogas to produce electricity with those of producing pipeline-grade methane. Electricity generation capacity is 500 kWe. Again, the value of the useable (marketable) energy produced is dependent on local conditions, so ranges of prices and revenues are shown.

Utilities may impose demand charges on customers with on-site power production, in order to accurately assign costs related to serving such customers [524]. Such charges are sometimes demand-based (maximum load during certain hours), while some are based on load and energy use [525]. The demand charge cost shown in **Figure 4-9 (b)** reflects a rate of $10/ kWe/ month applied to a 250 kWe load assumed for the customer. The gas pipeline cost shown in the figure reflects a pipeline 2 miles long, costing about $1/ GJ of methane marketed (see **Section 4.4.3**).

It must be stressed that results in **Figure 4-9** should not be viewed as a prediction of cash flows for an AD system. Rather, these costs and revenues are intended to provide insight into the relative magnitude of cash flows, and the potential for different revenue sources to support an AD installation. In practice, grants have supported digester systems, often covering 25% of the total system cost, along with a variety of revenue sources, including, but not limited to, energy sales, fiber recycling, fertilization improvements, and GHG credits. The net present value of various subsidies is compared in **Table 4-3**. A cash grant is consistently preferred over production incentives valued at 1.1 ¢/ kWh (as is available to open-loop biomass plants like digesters). Accelerated deprecation is valuable at higher discount rates, since the benefit is front-weighted. Zero-interest loans also outperform accelerated depreciation and the production incentive at discount rates of 4% and higher, but this subsidy is not likely to be funded going forward. A comparison of the PTC and cash grants was also carried out by Bolinger et al. [526], who found that for open-loop biomass systems, the cash grand was consistently preferred over the PTC.

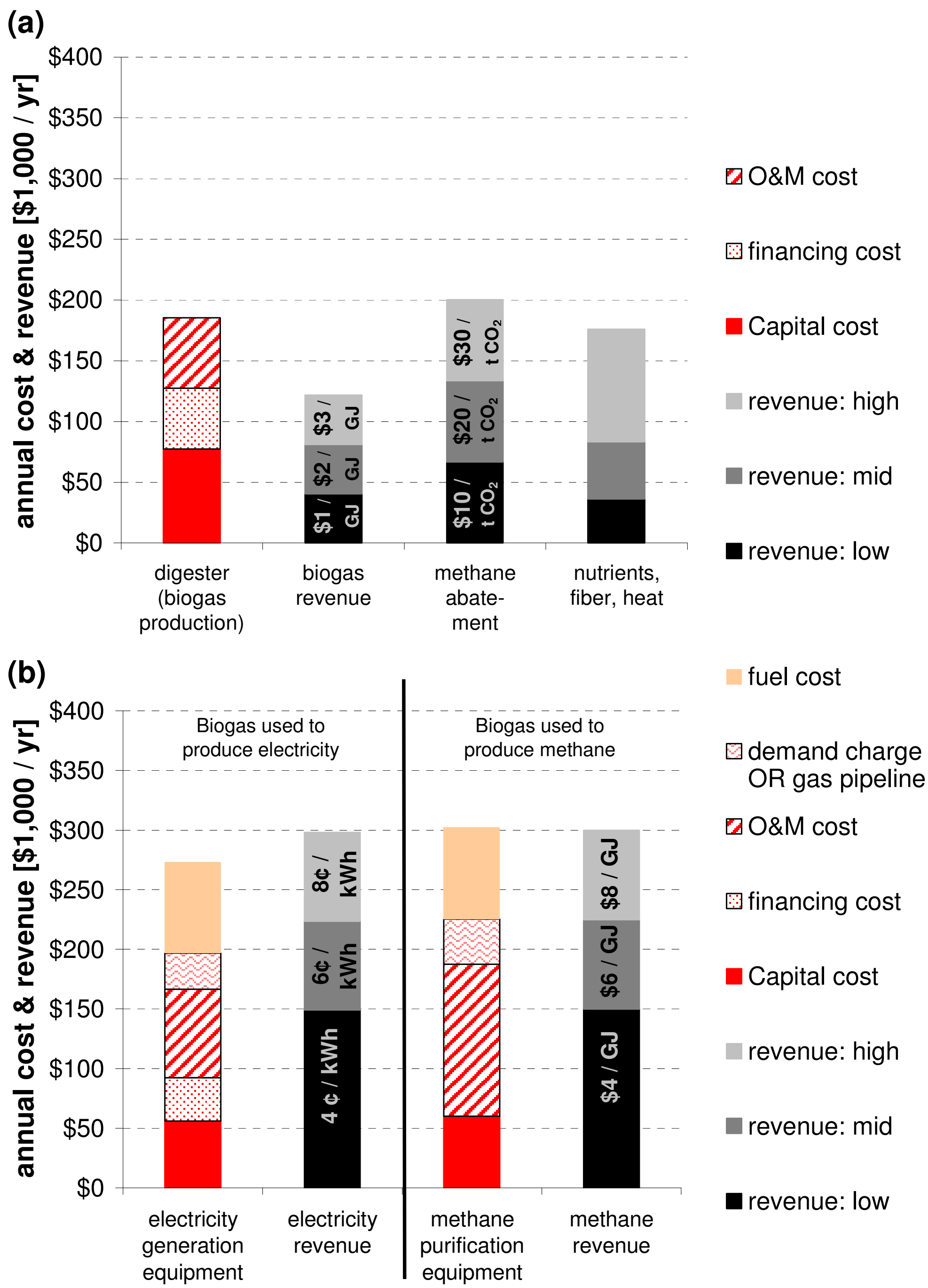

Figure 4-9. Comparing the economics of electricity and methane production from biogas: annual costs and revenues from a 1.4 MW thermal (HHV) biogas system. All capital costs are financed at 7% and amortized over 15 years. The top figure (a) shows annual costs and potential revenues of the biogas-producing digester. See text for details. The lower figure (b) compares annual cash flows for electricity and methane production. The "fuel cost" shown reflects a price of $2/GJ for biogas – this cost would be relevant if the biogas utilization equipment is managed separately from the digester. Cost for methane purification is from Figure 4-8, with breakdown between operation costs (O&M) and capital cost (including financing) from [520].

Table 4-3. Comparing the value of subsidies for anaerobic digester projects.

		¢/kWh[a] (0% DR)	¢/kWh (4% DR)	¢/kWh (7.5% DR)	¢/kWh (12% DR)
Baseline investment cost					
Capital cost of digester system	$4000/ kW	3.6	3.6	3.6	3.6
Value of subsidies					
Cash grant	25% of capital cost	0.9	0.9	0.9	0.9
Accelerated depreciation	5 years instead of 15	0.0	0.6	0.9	1.1
Production incentives (PTC or feed-in)	1.1 ¢/ kWh (10 year duration)	0.7	0.6	0.5	0.4
Zero interest loan	(first 10 years)	0.0	0.7	1.1	1.6

[a] Project lifetime of 15 years, 85% capacity factor; "DR" = discount rate

4.5 Costs & revenues of AD relative to those of animal products

Anaerobic digestion is largely a waste management strategy. In this section, the cost and revenues associated with building and operating an AD system are compared with costs of producing animal products on farms. Two representative farms are used in the comparisons. The first farm is a dairy with 1,000 milking cows and 450 replacement heifers. The second is a hog farm with 10,000 growing hogs and necessary reproducing sows.[8]

The 1,000 dairy cows together produce 26 t of milk per day (milk production per cow from [43]). Animals are exported from the farm as spent cows (454 kg live weight [LW] each day) and extra calves (85 kg LW per day). The value of spent cows is based on the value of chicken meat (food-energy content adjusted), while the value of extra calves is based on the value of beef. The hog farm produces 7900 kg LW per day from the growing hogs, along with 470 kg LW per day from reproducing animals sent to slaughter (mostly sows). Biogas is produced from manure on the dairy farm at a rate of 610 kW thermal (HHV), and from manure on the hog farm at 440 kW thermal.

[8] In practice, many farms specialize in the hog growing portion of the reproductive cycle ('finishing' phase), while others supply weaned piglets for growing. Many finishing farms produce in batches (all-in, all-out) to reduce disease transmission.

In **Table 4-4**, the economics of anaerobic digestion are shown relative to animal production economics, both from a cost perspective **(a)** and from a revenue perspective **(b)**. The values of products from the farms are described in **Table 4-9** in the Appendix. The cost of producing and flaring biogas is assumed to be $3.3 per GJ (as in **§4.4.3**). Note that in **Table 4-4** costs associated with animal products are higher than revenues; this is consistent with data reported by USDA, which show negative returns for animal product producers in many years. When a digester is constructed strictly for manure treatment and the biogas is flared, the calculated cost amounts to about 1% of the total cost of production for milk and hogs.

Table 4-4. Comparing costs (a) and revenues (b) of manure digestion with costs and revenues of producing animal products. The primary product is milk and pig meat for the dairy and hog farms, respectively. The total costs of milk and hog farms are from [527], while the value of animal products (AP) are from [43] (see Appendix for data).

(a) Cost-based

	Cost of animal farm	Cost of AD w/flaring	(as % of AP costs)
	[$ / day]	[$ / day]	[%]
Dairy	11,800	170	1.4%
Hog	11,300	130	1.2%

(b) Revenue-based

	Primary product	Primary + coproducts	Revenue from biogas	(as % of primary revenues)
	[$ / day] (primary products)	[$ / day] (all products)	[$ / day]	[%]
Dairy	8,900	9,400	210	2.4%
Hog	8,000	8,400	150	1.9%

4.6 NOx emissions and permitting

Burning a fuel often creates pollutants. The type and amount of pollutants released during biogas combustion depend on the chemical composition of the fuel, the type of power conversion device being used, along with associated exhaust treatment. Biogas often contains hydrogen sulfide (H_2S), and if biogas is not cleaned of H_2S before combustion, the exhaust will include sulfur aerosols (as SO_2).

Nitrogen oxides (NOx) are produced in high-temperature combustion processes, through the reaction of oxygen with nitrogen (both present in air). Stationary reciprocating engines can operate with low NOx emissions (less than 150 g NOx/ GJ fuel) [528-530]. On the other hand, in the absence of NOx emission controls, biogas engines can be tuned to increase efficiency at the expense of high NOx emissions [531]. NOx permitting is regionally specific, depending on compliance with ozone air quality standards.

Animal farms often have considerable emissions of volatile NH_3 from grazing, housing, and manure and feed management. The amount of volatilized N depends on farm-specific factors (animal types, manure handling and storage equipment and practices [470, 532] and diet [533]), as well as weather and climate. Annual emission factors are crude, but they are a convenient way to characterize the magnitude of ammonia emissions from farms. Based on animal N excretion from ASABE [415], ammonia emissions as a fraction of excreted N range from 15% for poultry manure managed in a wet form, to 18% for cattle, to almost 40% for broiler production (dry manure) [470, 532].

NOx emissions (modeled as NO_2 [534]) from energy conversion and N emissions from feed and manure management contribute to particulate matter air pollution ($PM_{2.5}$), but the degree to which reducing emissions of one or the other reduces $PM_{2.5}$ depends on the presence of other pollutants, and on meteorology [535].

For a representative farm digester system processing manure and producing about 500 kWe, NOx emissions from a reciprocating engine powering a generator amount to about 4.9 kg NOx-N/ day (based 2 g NO_2/ kWh electricity, adapted from [529-531]), while the manure management system (animal housing, manure storage, and application) likely emits about 180 kg N/ day (annual average, based on 900 kg manure N excreted/ day, and an ammonia emissions amounting to 20% of excreted N). Thus, the amount of N added to the atmosphere via biogas combustion is less than 3% of the N emitted from manure. If a digester system imports organic material, biogas production and resulting NOx emissions will be higher, but so likely will be the amount of N contained in the digested material.

4.7 Conclusion and Outlook

For manure management with AD, the digester (biogas-producing part of the system) is the base technology. It reduces the pollution potential (often measured in oxygen demand and microbe populations) of liquid and slurry manure, making low-odor separation and management possible. The improved hygienic nature of the solids makes fiber reuse as bedding more acceptable. The liquid portion is a good fertilizer, although the N,P,K ratios will depend on N retention (affected by storage practice and weather).

Improvements in animal genetics and feed quality could reduce the amount of manure excreted during production. For example, a genetically-modified pig with improved P-utilization and low manure P excretion has been created [536]. Such developments could reduce nutrient loads to soils and waterways, but manure management will probably be an important part of animal farming for some time. In addition, conveyer systems could enable relatively simple and inexpensive passive separation of solid and liquid fractions, whereby solids are composed, and liquids are used as high-quality fertilizer.

Digester cost in the future will be influenced by several factors, and it is not clear that capital costs will decrease. In particular, AD will note likely be widely applied on small animal farms since the evidence suggests small systems are prohibitively expensive. Also, smaller operations tend not to keep animals in confinement exclusively, so the supply of manure to the digester could be seasonal. Garrison and Richard report that AD systems are economic on relatively large hog operations (e.g. >5,000 head finisher operation), but also that increased energy prices and financial assistance are required to encourage adoption of manure-based AD systems [537]. The increasing size of digesters being installed worldwide, and the experience, knowledge, and increasing number of digester developers in the US in particular will likely tend to reduce capital costs of digesters. However, more advanced digester monitoring, which increases the reliability of digesters, will tend to add capital cots. Construction cost indices have been volatile as well. The sensitivity of electricity cost to changes in capital cost of digester systems is shown in the appendix, **§4.8.3**.

4.8 *Appendix*

4.8.1 Value of animal products (AP)

Table 4-5. Value of AP from [43] ('Quick Stats' and 'Agricultural Statistics, Annual'), in units as given (cwt = one hundred lb; LW = live weight).

	Milk	Chickens	Beef cattle	Pigs	Eggs	Turkey
	[$ / cwt]	[$ / lb LW]	[$ / cwt LW]	[$ / cwt LW]	[¢ / doz eggs]	[¢ / lb LW]
1990	13.7		74.6	53.7	83.5	
1991	12.3		72.7	49.1	78.5	
1992	13.2		71.3	41.6	66.3	
1993	12.8		72.6	45.2	18.2	
1994	13.0		66.7	39.9	17.6	
1995	12.8		61.8	40.5	18.0	
1996	14.8		58.7	51.9	20.2	
1997	13.4		63.1	52.9	23.1	39.9
1998	15.5		59.6	34.4	26.0	38.0
1999	14.4		63.4	30.3	28.6	40.8
2000	12.4	0.34	68.6	42.3	20.1	40.6
2001	15.0	0.39	71.3	44.4	23.5	39.0
2002	12.2	0.31	66.5	33.4	35.0	36.5
2003	12.6	0.35	79.7	37.2	46.3	36.1
2004	16.1	0.45	85.8	49.3	60.5	42.0
2005	15.2	0.44	89.7	50.2	67.0	44.9
2006	13.0	0.36	87.2	46.0	68.5	47.9
2007	19.2	0.44	89.9	46.6	64.0	
2008	18.5	0.46			79.1	

4.8.2 Manure properties & potential value

Table 4-6 describes manure production and properties. For animals involved in the milk- and egg-production systems, manure production is reported in units of 'manure per animal per day'. For animals grown for meat, manure production is reported as 'manure per finished animal' (ready for slaughter). The manure production for reproducing animals (including sows and beef cows) is also reported in units of 'manure per reproducing animal per day' [415].

Table 4-6. Manure production and energy yield parameters [43, 415, 538-540]. Fiber is assumed to be 30% of VS. "Total E content" represents the HHV of manure DM.

(a)

		Dairy cow	Dairy replace-ment	Hog (breed-ing)	Egg-Layer	Humans
Manure (excreted)	[kg / animal / day]	65.69	22.00	6.40	0.09	2.04
moisture	[mass fraction]	0.87	0.83	0.90	0.75	0.89
Dry matter (DM)	[kg / animal / day]	8.59	3.70	0.64	0.02	0.22
Volatile solids (VS)	[kg / animal / day]	7.25	3.20	0.56	0.02	0.13
Fiber	[kg / animal / day]	2.17	0.96	0.17	0.00	0.04
COD	[g /animal / day]	7815	3400	596	18	204
BOD	[g /animal / day]	1248	540	212	5	88
Nitrogen (N)	[g /animal / day]	433.08	120.00	42.60	1.60	13.62
Phosphorus (P)	[g /animal / day]	74.31	20.00	12.20	0.48	1.36
Potassium (K)	[g /animal / day]	106.46	28.65	28.20	0.58	4.77
Body mass	[kg / animal]	630.00	300.00	140.00	1.50	68.00
Ultimate CH4 yield	[l / kg VS]	150.00	150.00	350.00	350.00	350.00
	[% achieved]	0.95	0.95	0.95	0.95	0.95
Total E content	[MJ / kg DM]	16.01	16.01	15.27	14.50	11.11
Biogas E yield	[MJ / kg DM]	4.48	4.59	10.84	9.01	7.25

(b)

		Broiler chicken	Beef (on feedlot)	Hog (grower)	Turkey
Manure	[kg / finished animal]	36	4500	608	28.4
Moisture	[mass fraction]	0.74	0.92	0.9	0.74
DM	[kg / finished animal]	1.3	360	60.8	7.28
VS	[kg / finished animal]	0.95	290	49	5.84
Fiber	[kg / finished animal]	0.29	87.00	14.70	1.75
COD	[g / finished animal]	1050	300000	51400	6700
BOD	[g / finished animal]	300	67000	18500	1880
N	[g / finished animal]	53	25000	5110	434
P	[g / finished animal]	16	3300	828	125.6
K	[g / finished animal]	31	17100	2160	200
Body mass	[kg / animal] (avg)	1	400	70	6.8
Production duration	["finisher" days to slaughter]	45	155	156	115
Ultimate CH4 yield	[l / kg VS]	350	150	350	350
	[% achieved]	0.95	0.95	0.95	0.95
Total E content	[MJ / kg DM]	14.50	15.31	15.27	15.28
Biogas E yield	[MJ / kg DM]	9.05	4.28	9.98	9.94

Table 4-7. Representative value of nutrients and energy contained in manure. Part (a) of the table shows the value of nutrients and energy contained in one tonne of manure dry matter (DM). Value of nutrients is based on 50% of as-excreted nutrient content (see Table 4-6) and the fertilizer nutrient prices in part (b) of the table, which are averages of yrs 2001 through 2008 [541] (Nitrogen is average of ammonium and urea fertilizer prices).

(a) value of manure: nutrients, energy and fiber [$ / t manure DM]

	Milk	Chicken (meat)	Beef (finish stage)	Pig	Eggs	Turkey
Value of nutrients	28	33	47	53	54	46
Energy (as solid fuel)		44	46			46
Biogas energy	22	45	21	51	45	50
Fiber	5	4	5	5	4	5

(b) manure constituents

		Milk	Chicken (meat)	Beef (finish stage)	Pig	Eggs	Turkey
Nitrogen (N)	[kg N / t DM]	47.5	41.2	69.4	81.3	72.7	59.6
Phosphorus (P)	[kg P / t DM]	8.1	12.4	9.2	14.5	21.8	17.3
Potassium (K)	[kg K / t DM]	11.6	23.5	47.5	36.8	26.4	27.5
Fiber	[t fiber / t DM]	0.25	0.22	0.24	0.24	0.22	0.24
Energy (as solid fuel)	[GJ / t DM]	16.0	14.5	15.3	15.3	14.5	15.3
Energy (as biogas)	[GJ / t DM]	4.5	9.1	4.3	10.1	9.0	9.9

(c) value of manure components (all animals)

Nitrogen (N)	[$ / kg N]	0.70
Phosphorus (P)	[$ / kg P]	1.97
Potassium (K)	[$ / kg K]	0.56
Fiber	[$ / t DM]	20
Energy (as solid fuel)	[$ / GJ]	3
Biogas	[$ / GJ]	5

Table 4-8. Manure production parameters for representative large farms, compared to manure production by 20,000 humans. For example, a broiler operation growing 300,000 chickens produces 4.2 times the amount of phosphorus (P) than a town of 20,000 humans. These multiples include support animals (e.g. manure production for "Chicken (meat)" includes the laying stock required to produce broiler chicks; likewise, "Milk" includes heifers required to replace retired dairy cows). Calculated from Table 4-6 in appendix.

	Milk	Chicken (meat)	Beef (finish stage)	Pig	Eggs	Turkey	Human town
Number of producing animals	1,000	300,000	3,000	5,000	250,000	50,000	20,000
	[multiple of "human town" production]						
DM (dry matter)	2.3	2.0	1.6	0.5	1.2	0.7	1.0
VDM (volatile DM)	3.3	2.5	2.1	0.7	1.5	1.0	1.0
COD	2.3	1.8	1.4	0.5	1.1	0.7	1.0
BOD	0.8	1.2	0.7	0.4	0.7	0.5	1.0
N	1.8	1.4	1.8	0.7	1.5	0.7	1.0
P	3.1	4.2	2.3	1.2	4.5	2.1	1.0
K	1.3	2.3	3.5	0.9	1.5	1.0	1.0

Table 4-9. Supporting information for § 4.5, "Costs & revenues of AD relative to those of animal products" (values of various products from dairy and hog farms). The values of milk and meat are from Table 4-5. The value of extra calves is the value of cattle ready for slaughter; the value of "spent cows" is the same as that for chicken ($ / food calorie basis), converted to $ / kg LW. Values for biogas and electricity are assumptions, informed by Figure 4-11 and Figure 4-12.

Dairy farm			Hog farm		
Milk	[$ / kg]	0.35	Growers	[$ / kg LW]	1.01
Spent cows	[$ / Mcal]	0.54	Spent sows	[$ / kg LW]	1.01
	[$ / kg LW]	0.95			
Extra calves	[$ / kg LW]	1.91			
Biogas	[$ / GJ]	4	Biogas	[$ / GJ]	4
Electricity	[$ / kWhe]	0.06	Electricity	[$ / kWhe]	0.06

4.8.3 Electricity and natural gas

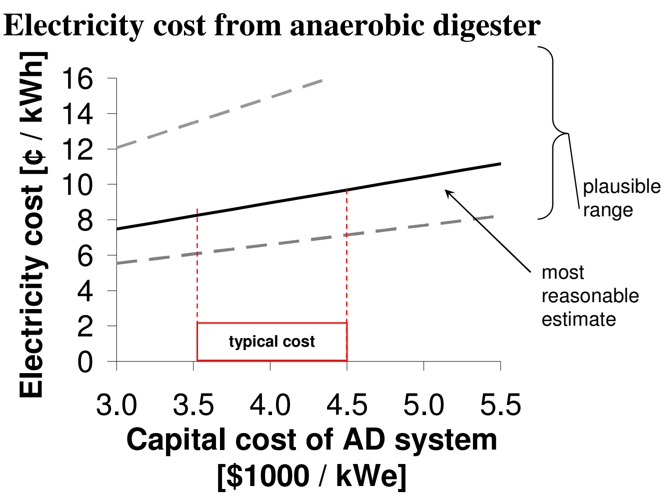

Figure 4-10. Sensitivity of electricity cost from an on-farm anaerobic digester system (cost includes both gas production and electricity generation equipment). Capital expenditure financed at 7%; project lifetime is 15 years; O&M cost for digester portion is 5% of digester capital expenditure annually (digester (gas-production) comprises 58% of total capital costs); O&M for the electricity generation is 2 ¢/kWh; capacity factor is 0.85.

Historic electricity prices:

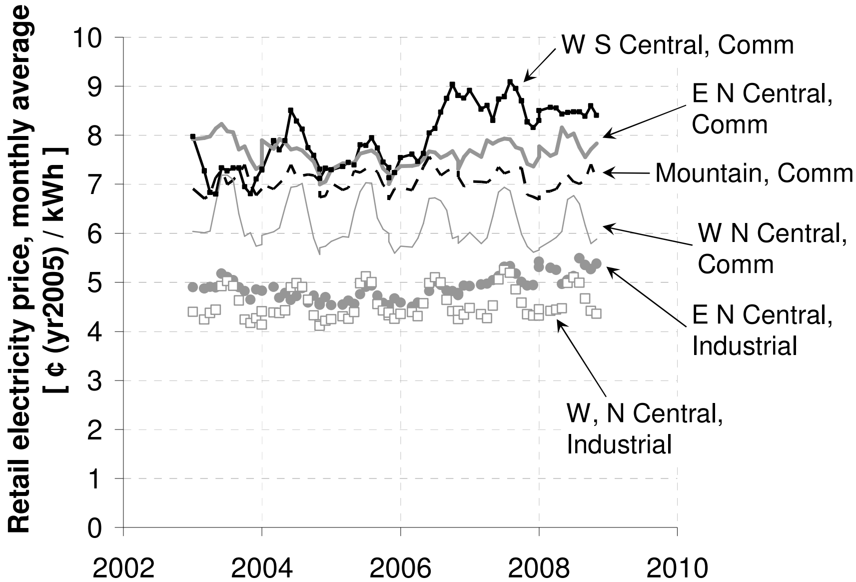

Figure 4-11. Cost of electricity in the US. Shows retail prices of commercial (lines) and industrial (dots) customers. The data are from EIA [542] and show average price paid by customers, including fees and charges (e.g. demand charges); adjusted to yr2005$ using CPI-U [76].

Historic natural gas prices:

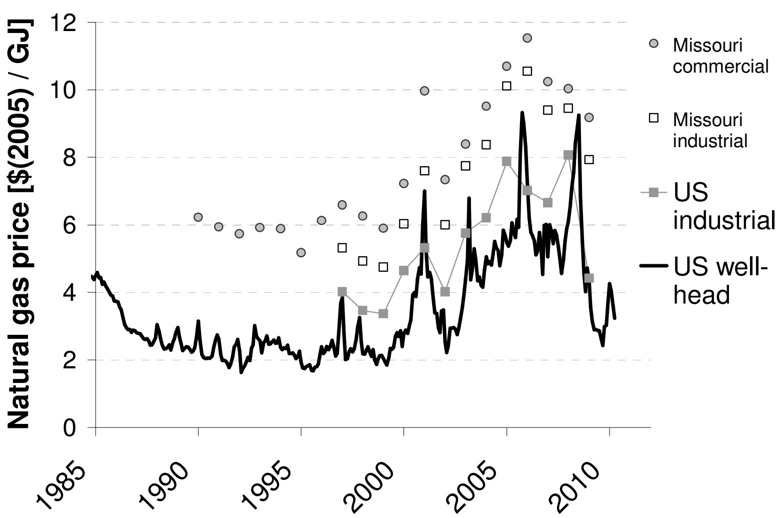

Figure 4-12. Natural gas prices in Unites States. [543]; adjusted to yr2005$ using CPI-U [76].

Natural gas pipelines:
Cost of natural gas pipeline from the digester to the pipeline ~150 k$ per mile [519, 520]. The map in **Figure 4-13** illustrates locations of pipelines in the US.

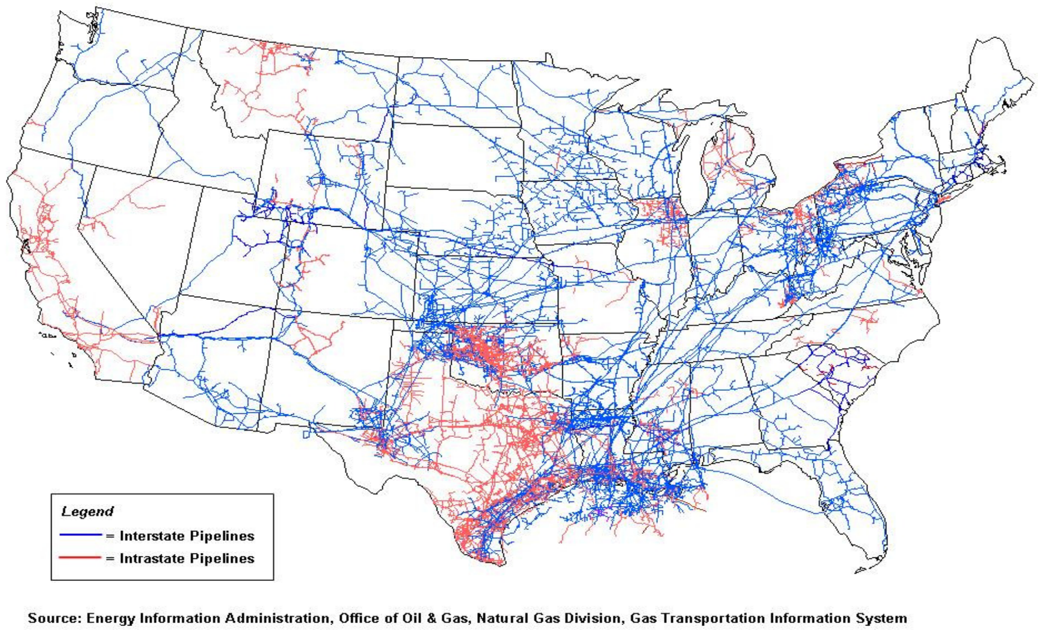

Figure 4-13. Map of natural gas pipelines in the US [544]. Dark lines are interstate; light lines are intrastate.

5 Life cycle assessment of manure management with energy harvesting

Abstract. A life cycle assessment of the greenhouse gas (GHG) emissions from manure management with energy harvesting is presented, and these results are compared to the overall impact of producing animal products. The magnitude of GHG reductions achieved with anaerobic digestion (AD) depends heavily on baseline methane (CH_4) emissions from manure management, and whether the comparatively lower CH_4 emissions from manure management with AD are considered to be a consequence of installing the AD system. When baseline CH_4 emissions from manure storage are substantial, and fugitive emissions from the energy-harvesting scenario are low, the cost of reducing GHG emissions via AD is below $20 per tonne CO_2eq. On the other hand, if the energy harvesting scenario is not responsible for manure emissions abatement, and fugitive emissions are high from the digester (7% of CH_4 produced), the biogas produced has about the same GHG impact as fossil natural gas. If CH_4 emissions from manure are eliminated as a consequence of energy harvesting, 45% of the total GHG impact of producing pigs is eliminated. The CH_4 emission reduction from manure storage is responsible for most of this reduction. The overall impacts of milk, chicken (meat) and turkey are reduced by about 30%, while the impact of the beef production system is reduced only slightly.

5.1 Introduction

Animal manure can be a resource that provides value (e.g. via nutrients for fertilization, fiber for bedding, energy), or it can be a waste in need of treatment [545]. When animals graze, manure is not usually a pollutant, since it is deposited directly on the land and is recycled. Grazing animals which congregate near surface water would be an exception, whereby manure deposited into or near the water may be a pollution concern. Likewise, manure deposition may be concentrated where grazing animals are offered supplementary feed.

On concentrated animal feeding operations (CAFOs), manure is deposited onto dirt, cement, or slatted surfaces, and must be managed. The availability of land on which to spread manure and the transport required to deliver manure to the land are important factors influencing the value of manure from CAFOs. However, manure management is just one aspect in the economic decisions of animal producers. Large farms may be more economic than smaller ones, even if manure is not useful and is managed at a cost [546].

Greenhouse gas (GHG) emissions from manure occur during animal housing [331, 547], liquid manure storage [548], solid manure storage and composting [549-551], and after field application [359]. Manure management and land application practices affect emissions [552], as do temperature and water content. Thus, emissions associated with manure must be determined for particular practices and local ecological conditions [553].

Anaerobic digestion (AD) is a useful way to obtain value from organics [554, 555]. Non-AD technologies also exist for harvesting energy from manure, including combustion and gasification [490]. Recycling biomass (also called “cascading” [14, 236]) can reduce pressure on land resources that would otherwise be called into service were it not for the reuse [238]. In addition, when energy is harvested from manure using AD, nutrients are retained in the digested material, preserving the manure’s potential to be used as fertilizer.

This chapter provides a life cycle inventory of the GHG emissions associated with using manure to power (MtoP) technologies to process animal manure. After a literature review, the system boundaries and functional unit are defined. The inventory is then presented, followed by results and discussion of changes from the baseline scenario. The chapter concludes with an estimate of the degree to which MtoP schemes could offset the GHG impact of animal products. The aim of characterizing losses from manure management is to inform an assessment of the impacts of integrating biofuel production with animal production (**Chapter 6**).

5.2 Literature review: LCA of energy-harvesting

A few life cycle assessments (LCA) of MtoP have been performed. Reijnders and Huijbregts compute the life cycle GHG emissions associated with burning animal wastes in the EU, using both energy-based and economic-value-based allocation of impacts associated with manure production [363]. This study highlights the impact of the allocation method used in the LCA. Berglund and Borjesson completed a life cycle assessment of energy used for biogas production [556], as well as the GHG balance of biogas facilities [280] and the magnitude of offsets produced [557], all under Swedish conditions. They find the fuel cycle emissions of biogas are between 10 and 20 g CO_2eq per MJ of biogas.

Ghafoori et al. study the life cycle GHG impact of a beef feedlot in Canada, with and without AD, and report that the emissions reductions associated with offsetting electricity production dominate the comparative result [558]. Ghafoori et al. also report that GHG credit prices as high as \$125/ t CO_2 would not make the AD system economic [559], while noting that non-GHG-related motivations may provide incentive for AD installation [560].

Wulf et al. describe how AD system design can influence life cycle GHG emissions. Covering the digested slurry storage tank reduces post-digestion methane (CH_4) emissions at a cost of about \$30/ t CO_2. Injecting slurry into the soil reduces nitrogen leakage and subsequent nitrous oxide emissions, at a cost of about \$50/ t CO_2eq. Adding the organic municipal waste increases yield of

CH_4 from manure-based digesters, but is not likely a cost-effective solution compared with composting these non-slurry organics [561].

Sandars et al. show that AD (without fugitive emissions) followed by injecting digested slurry into soil results in the lowest environmental impact among manure management options [552]. Other management techniques (e.g. more frequent manure collection from animal housing and slurry separation followed by energy recovery via incineration) can reduce GHG emissions from manure management, but regional differences in climate and production practices also effect GHG emissions [553].

The work presented here builds upon this body of research by estimating the consequential impacts of installing MtoP on animal farms in the US. Such a focus is warranted by the increasing number of MtoP installations and the potential importance of revenue from GHG credits (outlined in **Chapter 4**).

5.3 System boundary and consequential impacts

In a consequential LCA of MtoP, the perspective taken regarding business as usual (BAU) manure management is a very important consideration. In the case of animal farms, CH_4 from liquid manure storage is a substantial portion of the GHG impacts of some animal products (particularly for hog production, see **Chapter 3**). If adopting a particular manure management practice (e.g. AD) is deemed to be responsible for reduced CH_4 emissions from manure storage, the adoption decision can be credited with the corresponding GHG reductions. Protocols have been developed to try to determine the CH_4 emissions reduction associated with a particular AD installation by estimating what type of manure management would be in place if not for the digester (e.g. [562]).

In this LCA, the system boundary includes only the manure management system. Manure enters the boundary when it is voided by an animal in confinement. The manure can exit the system boundary in a number of ways. Some nutrients may leave the manure management system via volatilization, leaching and runoff, or nutrients may be exported to a crop production system via land application. While the adoption of a manure processing scheme can influence nutrient availability and cycling when applied to land [563], the degree to which excreted manure nutrients are put to use as fertilizer depends on specific practices. The focus of the present chapter is on the GHG balance of the manure management system, and so the flows of nutrients entering and leaving the system are considered, but the impacts of utilizing the manure as fertilizer are not, since the utilization of manure nutrients depends on the locations of both the CAFO and crop production.

5.4 Life cycle inventory

Impacts associated with manure management include emissions from fuel and infrastructure via transport of manure to a processing facility (if necessary), construction and maintenance of the energy harvesting facility and associated equipment, as well as any fuel and electricity used during startup and operation. In addition, CH_4 and nitrous oxide (N_2O) can be emitted during manure management. In the following paragraphs I will utilize the framework outlined in **Chapter 2** to calculate the GHG impacts associated with the manure byproduct system:

1. **Feedstock acquisition**
2. **Transport** from point of generation to point of processing, and from processing facility to point of use or storage
3. **Processing** the byproduct
4. **Use** of the feedstock or products from processing
5. **Storage** and sequestration of biomass C

Two systems are described. One is a business-as-usual (BAU) scenario representing average current practices, while the other is an energy-harvesting (E-harvesting) scenario where some of the energy of the excreted manure is exported from the manure management system.

5.4.1 Feedstock procurement

The manure is received into the manure management system upon being voided by the animal. No impacts from manure production (i.e. animal farming) are assigned to subsequent manure utilization. Put another way, the excreted manure carries no environmental burdens into the manure management system.

The properties of fresh manure are listed in **Table 5-1**. When beef cattle excrete manure, about 8% of the manure mass is dry matter (DM, i.e. not water), but when collected from feedlots, much of the moisture may have evaporated, and the DM content of the manure can be as high as 90% by weight. Of particular concern for manure utilization as a solid fuel, the DM collected from beef cattle feedlots is often 50% ash by weight due to dirt incorporation. Alternative practices, such as covering feedlot surfaces with coal fly-ash can improve the quality of collected manure [489]. The manure moisture contents assumed for the BAU and E-harvest scenarios are shown in **Table 5-2**, and are used to calculate impacts of manure transport to processing (where applicable).

Table 5-1. Manure properties as collected. Sources: [10, 415, 485, 538, 540, 564-567]. "HHV" = higher heating value. "B_0" = ultimate methane yield.

		Dairy cattle	Chickens (meat)	Beef (on feedlot)	Beef (not on feed)	Pigs	Eggs (for eating)	Turkeys (meat)
DM	[g DM/ g manure]	0.14	0.27	0.08	0.12	0.10	0.25	0.26
VS	[g VS/ g DM]	0.85	0.73	0.81	0.89	0.82	0.73	0.80
Ash	[g ash/ g DM]	0.15	0.27	0.19	0.11	0.18	0.27	0.20
E	[MJ/ kg DM]	16.9	14.6	16.1	17.9	16.3	14.5	16.0
B_0	[L CH_4/ kg VS]	170	350	200	150	338	350	350

Table 5-2. Manure moisture content (mass basis) as voided [415], and as collected for the BAU scenarios and E-harvesting scenario. NC = 'not collected'

	Dairy cattle	Chickens (meat)	Beef (on feedlot)	Beef (not on feed)	Pigs	Eggs (for eating)	Turkeys (meat)
as voided	0.86	0.73	0.92	0.88	0.90	0.75	0.74
as-collected, BAU	0.87	0.50	0.50	NC	0.93	0.75	0.50
as-collected, E-harvest, AD	0.87	0.50	0.71	NC	0.93	0.90	0.50

5.4.2 Transport

Manure collected on a farm may be used on-site, or may be transported from the point of excretion to a facility for centralized processing. I assume that feedstocks being processed off-farm travel 50 km by truck (two-way transport for high-moisture feedstocks being digested at a central facility and one-way transport for dry residues that are combusted). Facanha and Horvath report that long-haul truck freight is associated with 122 g CO_2eq per tonne-km (tkm), including infrastructure impacts [155]. Since it is unlikely that manure-hauling trucks operate with the efficiency of long-haul trucks, it is assumed that trucking manure emits 180 g CO_2eq per tkm (as in [158]). The emissions associated with trucked manure depend on the moisture content, and range from 1.5 g CO_2eq/ kg DM/ km for fresh feedlot beef manure to 0.45 for broiler chicken litter (**Figure 5-1**).

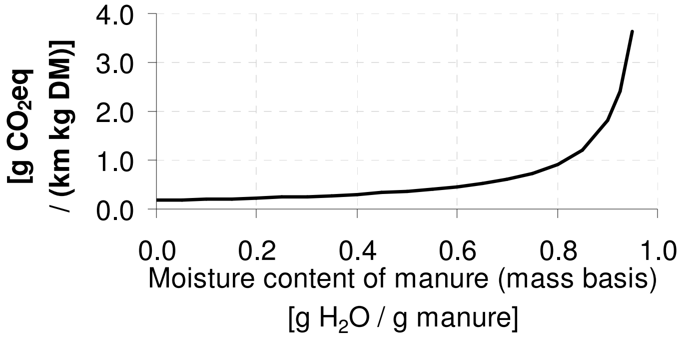

Figure 5-1. GHG impact of transporting manure, as affected by moisture (H_2O) content of the manure to be transported. Note that dry matter (DM) content of manure = (1 – moisture content).

5.4.3 Processing and storage

With manure management, manure storage may occur in-place during accumulation, as well as after processing. Furthermore, practices such as heap composting may be considered simultaneous processing and storage. Thus, the processing and storage stages of the LCA framework are discussed together here.

During manure accumulation and management, some of the carbon (C) and nitrogen (N) contained in excreted manure is volatilized. When manure is allowed to accumulate over extended periods in an aerobic environment (with oxygen), C in manure is oxidized (either biologically or thermo-chemically). Likewise, organic C in an anaerobic state (without oxygen) is converted to CH_4 and CO_2 by microbes. During composting, the aerobic conversion of manure organic compounds can be vigorous and produce high temperatures within the heap. If manure is collected frequently (as is often the case when manure is handled as a liquid), the liquid can be pumped to a processing facility immediately, so C losses during collection are minimal. The C- and energy-loss factors are assumed to be represented by the same value, shown in **Table 5-3** for the BAU and E-harvesting scenarios.

When manure is managed and stored as a liquid or slurry, anaerobic conditions may lead to CH_4 production and emission. The maximum amount of CH_4 that can be generated from a particular organic material under anaerobic conditions is indicated by its ultimate methane yield (B_0). Without CH_4 collection, a substantial fraction of B_0 can be released to the atmosphere during anaerobic manure storage. The magnitude of CH_4 emissions from manure (as reported by the EPA [102]) amount to 65% of the B_0 excreted for hog-related manure production, 30% for milk-related manure production, and 20% for egg-related production (**Table 5-3**).

Methane emissions during manure storage are largely eliminated for the AD case. However, fugitive CH_4 emissions may occur from the digester and processing equipment, and from post-digestion manure storage. Such fugitive emissions have a strong temperature dependency, and can be relatively minor [568] or more substantial [436, 548, 569]. For this inventory, a fugitive emission factor of 2.5% of the ultimate methane yield of the material entering the digester is assumed.

Transformations of manure N also occur during accumulation and storage. Excreted urea may quickly be converted to ammonia. As with carbon, N may exist in a reduced form (e.g. NH_4^+) or an oxidized form (e.g. NO_3^-). Biologically-induced transformations are often dependent on moisture and oxygen levels. Volatilization of ammonia from manure will depend on pH and temperature, as well as the nature of the surface on which manure is deposited. During management, conditions including moisture content of the manure are variable. Because of this diversity, modeling N transformations is outside the scope of the current work. Rather, a parameter is estimated which represents the fraction of excreted manure N that is lost to the environment during manure management (that is, the fraction of N that is not used as a soil amendment or other product outside of the manure management system boundary). The N loss factors in **Table 5-3** are chosen based on literature values [570-573]

The emission of nitrous oxide (N_2O) from manure management is system-dependent, and the fraction of excreted N emitted as N_2O from management system components can range from 0% for a fully anaerobic lagoon to 10% for intensive composting [221].

For this assessment, 2% of excreted animal manure N is assumed to be emitted as N_2O [146]. This is a "lifetime" emission factor, meaning that of 100 g N excreted by an animal, 2 g N is eventually emitted as N_2O; these emissions may occur within or outside the system boundary defined here. The N loss parameter is applied to estimate the fraction of the lifetime manure N_2O emission that arises during manure management. For example, if 60% of the excreted manure N is lost during management, then this same percentage of the lifetime N_2O emission is assumed to be emitted during management. The N_2O emission factors for cattle manure management (i.e. the assumed N-loss in **Table 5-3**) agree with emission factors calculated from EPA estimates (dairy and beef feedlot, see appendix, **§5.7**). The BAU N_2O emission factor for poultry and hog manure (1% of excreted N lost as N_2O during management, **Table 5-3**) is higher than the 0.5% calculated from EPA data. These higher values are adopted, however, because N loss from poultry and pig operations is relatively high [570, 571], periodic sampling from manure packs may underestimate emissions [353], and N lost from the farm system can be subsequently converted to N_2O [147].

Using manure as an energy feedstock might increase frequency of manure collection at feedlots, which might, in turn, reduce the fraction of excreted DM and N lost during manure management. The N losses from the E-harvesting scenarios are assumed to be half of the BAU loss factors (**Table 5-3**).

Undoubtedly, the loss factors estimated in **Table 5-3** are crude estimates based on a wide range of plausible values. Thus, they should not be applied to any particular farm without confirming their applicability.

Table 5-3. Dry matter (DM), energy and carbon (C) retention, nitrogen (N) loss, methane emission, and N_2O emission factors assumed for the business-as-usual (BAU) and energy harvest (E-harvest) scenarios. Units = [fraction of excreted]. Methane emission factors indicate the fraction of the ultimate methane yield (B_0) emitted during manure management, and is calculated using total CH_4 emissions from manure management estimated by the EPA [102] and the total methane potential of manure in the US (Chapter 4, Table 4-2).

		Dairy cattle	Chickens (meat)	Beef cattle (on feed)	Pigs	Egg-producing chickens	Turkeys
DM loss	BAU	0.08	0.18	0.52	0.24	0.15	0.20
	E harvest	0.04	0.09	0.26	0.12	0.07	0.10
Energy, VS and C loss	BAU	0.10	0.25	0.65	0.30	0.20	0.25
	E harvest	0.05	0.13	0.33	0.15	0.10	0.13
N loss	BAU	0.25	0.50	0.86	0.50	0.50	0.50
	E harvest	0.13	0.25	0.43	0.25	0.25	0.25
Methane (fraction of Bo)	BAU	0.28	0.02	0.02	0.66	0.23	0.02
	E harvest	(fugitive, see text)					
N_2O emission (as N)	BAU	0.005	0.010	0.017	0.010	0.010	0.010
	E harvest	0.003	0.005	0.009	0.005	0.005	0.005

In addition to emissions from manure, GHG emissions occur during the construction, maintenance and operation of manure management equipment. The impacts of basic manure management infrastructure required for the BAU and E-harvest scenarios were included with animal farm operation (see **Chapter 3**) and are not considered separately here. The E-harvest scenario, however, requires additional equipment.

Construction impacts of the E-harvesting facility are estimated using the Economic Input-Output LCA (EIOLCA) tool [397]. For the digester and methane production facilities, the "Other nonresidential structures" sector in EIOLCA is used (612 g CO_2eq / $ [purchaser price], yr2002 $), while the "Motor and generator manufacturing" sector (582 g CO_2eq / $ [purchaser price]) is used for the electricity generation equipment. Capital and maintenance costs from **Chapter 4** are adjusted to yr2002$ using the CPI [76].

An E-harvesting facility uses fuel and electricity during operation. Biogas energy facilities can be designed to supply biogas, methane or electricity energy products and services. The operation of a slurry-based biogas plant requires 300 MJ electricity (MJe) per tonne of DM processed, about 5% of the gross methane energy output [280]. Solid material added to a digester (e.g. cattle manure collected from feedlots) requires an additional 30 MJe per tonne DM for processing [574]. When a biogas facility is producing electricity, these on-site demands may be subtracted from the gross electricity production. However, a biogas facility producing CH_4 (not electricity) will have to import electric power. The GHG intensity of electricity in the US averages 180 g CO_2eq/ MJ (650 g CO_2eq/ kWh), but is quite variable regionally, and emissions tend to be higher in the Midwest [394]. In this case, electricity required is assumed to have a GHG impact equal to the US average.

5.4.4 Use

When biogas is used to produce electricity, the generation efficiency is assumed to be 35% (HHV). For biogas upgrading to methane, CO_2 removal and methane compression each require 0.35 kWh(e) per cubic meter of CH_4 produced [574], which amounts to approximately 8% of the energy content of the methane for each. The GHG impacts of natural gas offsets are 60 g CO_2eq/

MJ (HHV) (as in [123]. As discussed above, the GHG intensity of electricity is regionally variable, and is particularly high in parts of the Midwest [394]. Animal production in the US does have a concentration in the Upper Midwest, but there are also clusters of production in the Southeast, Mid-Atlantic and the West [575]. Thus, the GHG intensity of electricity offsets is assumed to be equal to the US average (180 g CO_2eq/ MJ). For cases where the electricity that is offset has higher than average emissions, the GHG reduction associated with the electricity produced from manure will be correspondingly higher.

5.5 Results

The GWP of manure management is reported for a BAU scenario – meant to represent current animal production practice in the US – and an energy harvesting (E-harvest) scenario, where electricity is produced using either anaerobic digestion (for wet manure) or combustion (for dryer manure).

The results in **Figure 5-2** show impacts of manure management per kg of manure dry matter excreted by the animal (kg DMx). The biogas systems reduce CH_4 emissions from hog, egg and dairy production, where manure is routinely stored in lagoons, tanks, or pits. The electricity production is higher for chickens (meat) and turkeys because this manure is collected as a relatively dry pack and combusted. During AD, in contrast, not all organic matter is harnessed for energy, since only 30-60% of the heat content of the manure is converted to biogas. With MtoP, manure management can be a net GHG sink, if electricity exported from the manure management system is credited to offset grid electricity.

In order to assess the magnitude of emissions reductions associated with MtoP, **Figure 5-3** shows all impacts associated with animal production (as assessed in **Chapter 3**), along with the emissions reductions from MtoP (including electricity production credits). About one third of the emissions associated with milk, chicken, egg and turkey production can be eliminated or offset by harvesting energy from manure. For hogs, the relatively large magnitude of manure CH_4 reductions, along with electricity credits reduce total emissions by 45% compared to BAU. For beef cattle on feedlots, manure CH_4 is a relatively small emission in the BAU scenario, and the total emission reduction is 12% of feedlot impacts. When impacts from beef cattle on the range are included, the E-harvesting from manure at the feedlot reduces total beef impacts by only 3%. The manure from beef cattle not on feed (on the range) is not available for energy harvest, so MtoP is not an option for these operations.

The results discussed above make it clear that the magnitude of CH_4 emissions during manure management in the BAU scenario has a large effect on the GHG impact of E-harvesting. At the same time, fugitive CH_4 emissions from E-harvesting facilities can reduce the effectiveness of GHG abatement from AD. **Figure 5-4** shows the sensitivity of the GHG impact of biogas to the BAU manure CH_4 emission, and the fugitive emission from the AD facility. If there are no CH_4 emissions from the BAU manure management, and no fugitive emissions during E-harvesting, the only life cycle impacts are from building and operating the digester, estimated at 10 g CO_2eq/ MJ (HHV) of biogas collected. On the other hand, if there are no BAU CH_4 emissions from manure, but fugitive emissions amount to about 7% of the ultimate methane yield (B_0), then the GHG impact of the biogas produced is about the same as fossil natural gas (HHV basis). Along these same lines, the BAU CH_4 emission from manure is a major determinant of the cost of GHG abatement with CH_4 harvesting. If the baseline CH_4 emission from manure is substantial, digesting the manure and destroying the CH_4 offers GHG abatement at costs lower than $20 per tonne CO_2eq (**Figure 5-5**).

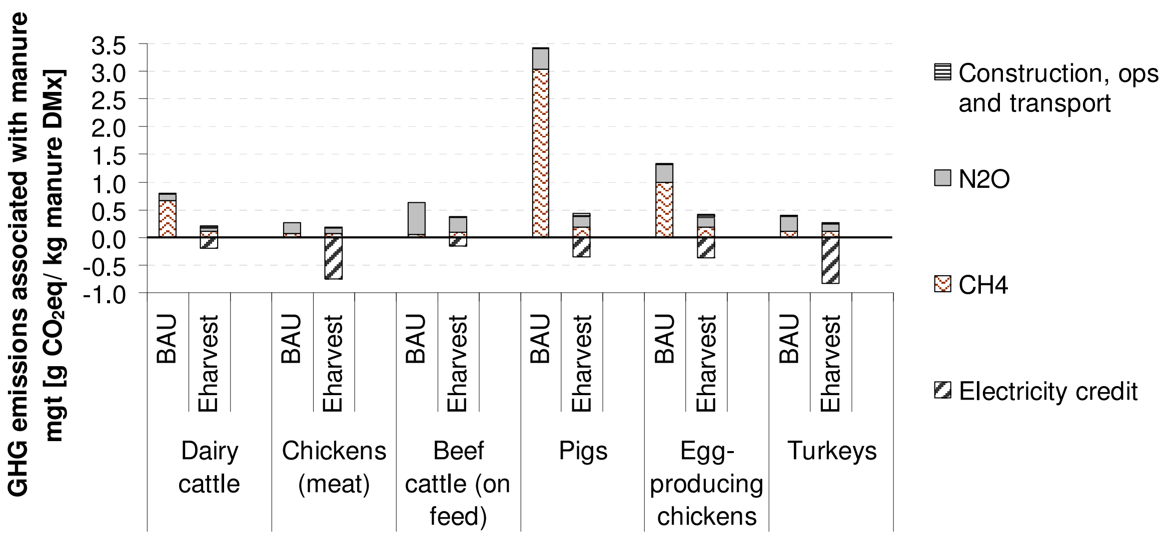

Figure 5-2. GHG impact associated with managing manure on animal farms, shown in units of g CO_2eq per kg of manure dry matter excreted by the animal (DMx). For pig, dairy and egg-related manure, the energy harvesting (Eharvest) scenario lowers emissions of CH_4 from manure storage. E-harvesting scenarios are credited with 180 g CO_2eq per MJ electricity exported to the grid.

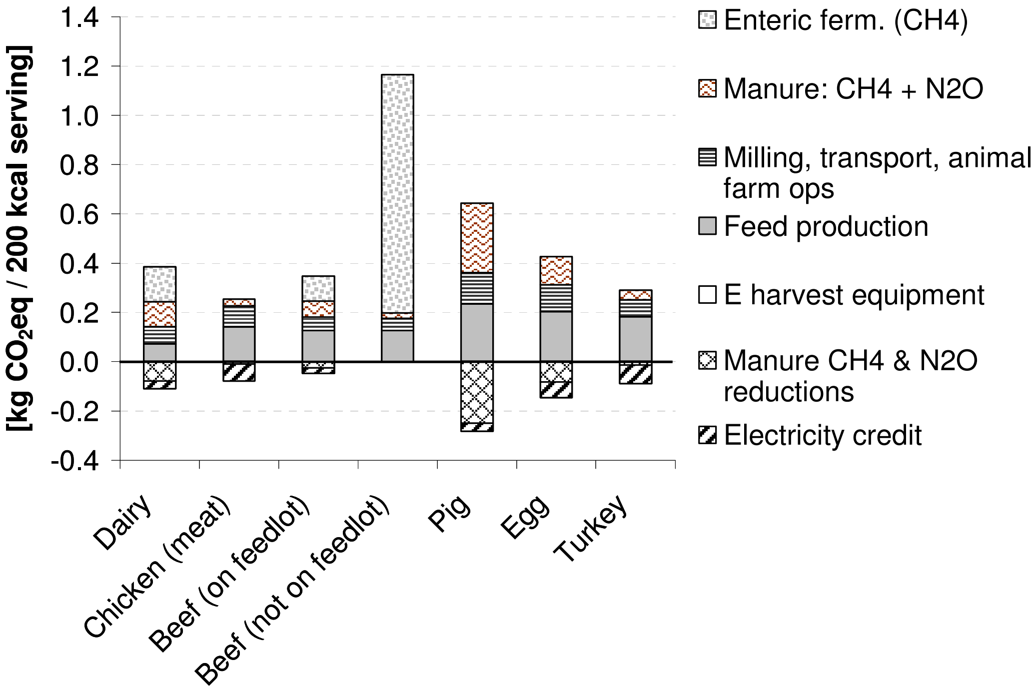

Figure 5-3. GHG reductions and offsets from harvesting energy from animal manure, compared with total BAU impact of animal production. For reference, the figure shows Baseline emissions associated with manure management, as well as the CH_4 and N_2O reductions resulting from improved manure management with manure to power (MtoP). Electricity production is credited with 180 g CO_2eq per MJ electricity exported to the grid. The "E harvest equipment" is an additional (small magnitude) emission resulting from construction and maintenance of the MtoP facility.

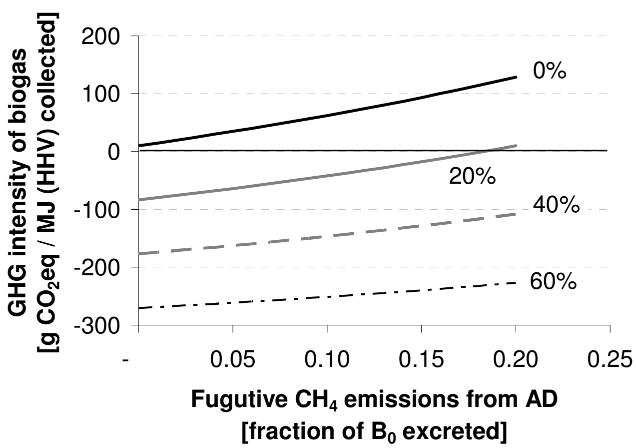

Figure 5-4. GHG intensity of biogas from anaerobic digestion (AD) facilities. X-axis: Fraction of the ultimate CH_4 yield (B_0) that is produced but not collected with the AD system. The lines represent different "business-as-usual" methane emission scenarios, labeled with the percent (%) of B_0 that would have been emitted from manure without the AD system. For the lines labeled with a non-zero percentage, the AD system is responsible for offsetting CH_4 emissions from manure management.

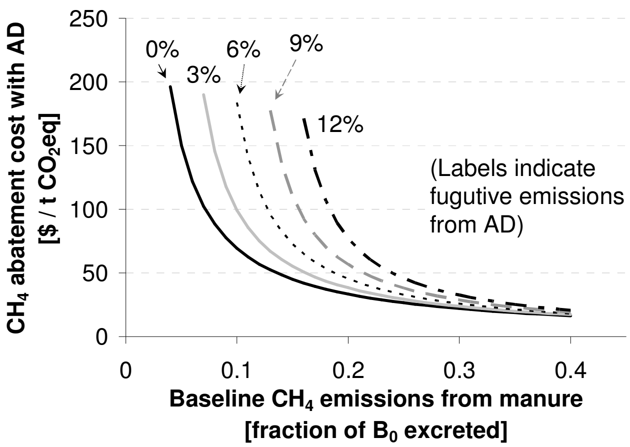

Figure 5-5. Sensitivity of cost of GHG abatement associated with biogas capture. These results only include costs and impacts of biogas production and flaring, and do not include E-harvesting equipment (the only GHG abatement represented is from reducing CH_4 emissions from manure). The vertical asymptote occurs where the baseline CH_4 emission from manure is equal to the fugitive emission from the CH_4 capture scenario (i.e. as net GHG abatement approaches zero).

5.6 Conclusion

This LCA focuses on the manure management phase of animal production and compares the GHG impact of harvesting energy from manure with the total life cycle emissions of producing animal products. The assessment includes construction and operation of energy harvesting infrastructure, as well as transport of manure. Emissions reductions achieved as a consequence of energy harvesting are included. However, this assessment excludes changes in emissions during the use of manure as a soil amendment. Results indicate that methane emissions reductions and GHG credits from electricity production can reduce impacts of animal production by 3-45%. For pig production, manure CH_4 reductions are high, so the E-harvesting scenario reduces total impact by 45%. At the other end of the range, a large fraction of the GHG impact of beef is from enteric fermentation, which is not affected by E-harvesting, so reductions are only 3%. Emission credits from electricity production from chicken (for meat) and turkey manure can lower overall impacts by 30%. Biogas harvesting is shown to be an inexpensive GHG abatement strategy, given that installing the digester is responsible for reducing CH_4 emissions from manure in the baseline case.

5.7 Appendix

N_2O emission factors

To estimate the fraction of excreted manure N emitted as N_2O during manure management, I have divided the total N_2O emissions for the animal categories estimated by the EPA [221] by the total amount of excreted manure N (calculated in **Chapter 3**). As shown in **Table 5-4**, the EPA N_2O emissions imply that N_2O emission factors during manure management are between 0.5% and 1.7%.

Table 5-4. N_2O emissions from manure management. The emissions factors [% of excreted N] are calculated using the N_2O emissions from EPA [Gg N_2O (N basis) / yr] [221] and the manure N excretion calculated using factors in Chapter 3 [Gg N excreted / yr].

	(year)	Dairy cattle	Chickens (meat)	Beef (on feedlot)	Beef (not on feed)	Pigs	Eggs (for eating)	Turkeys (meat)
N_2O emissions from manure management		[% of excreted N]						
	2004	0.47%	0.41%	1.86%	0.00%	0.50%	0.53%	0.52%
	2005	0.48%	0.42%	1.90%	0.00%	0.50%	0.53%	0.53%
	2006	0.48%	0.42%	2.00%	0.00%	0.51%	0.52%	0.51%
	2007	0.49%	0.42%	2.05%	0.00%	0.51%	0.53%	0.49%
		[Gg N_2O (N basis) / yr]						
	2004	7.5	2.0	13.2	0.0	3.1	0.9	0.6
	2005	7.6	2.1	13.3	0.0	3.1	0.9	0.6
	2006	7.8	2.1	13.7	0.0	3.1	0.9	0.6
	2007	8.0	2.1	13.7	0.0	3.2	0.9	0.6
Manure N excretion		[Gg N / yr]						
	2004	1592	496	711	2281	605	167	111
	2005	1603	503	702	2287	607	168	108
	2006	1623	504	687	2288	614	170	111
	2007	1634	505	672	2283	638	168	116

6 The ongoing integration of fodder and biofuel markets

Abstract. In yr2009, the US was the largest fuel ethanol (EtOH) producer in the world, making 60% more than Brazil, the next largest producer. EtOH supplied 4% of liquid fuel energy (HHV) in the US transport sector - up from 1% in yr2003. The mass of co-product feed produced from corn EtOH in the US is greater than the mass of soybean meal fed to animals, which indicates the strong interaction between biofuel production and animal production. A life cycle assessment of an integrated biofuel and animal production facility is performed. A large, mixed animal farm is modeled which consumes co-product feed from biofuel production. Manure from the animals is collected and energy is harvested, providing heat and electricity for EtOH manufacture and some electricity for export. By using the low-GHG fuel harvested from manure, EtOH is produced with life cycle GHG impact of 40 g CO_2eq per MJ (not including any land use change impacts). Overall GHG emissions are 28% lower from the integrated facility than from the two separate systems, and reductions from the animal sub-system make up 70% of gross reductions. If EtOH is credited with all emissions reductions (including those from animal production), the fuel has a negative GHG impact. I conclude with a hypothetical scenario in which cellulosic biofuel policies cause animal production to shift away from beef to less GHG-intensive meat. If biofuels are assigned responsibility for this shift, the GHG benefit would be about equal to the per-MJ co-product feed credit now typically assigned to corn grain EtOH.

6.1 Introduction

The energy and agricultural markets are becoming integrated [202] and increasing demand for biomass (through government mandates and financial incentives) could induce changes in land use and biomass markets (see discussion of land use in **Chapter 1**). The current patterns and magnitudes of fodder biomass use are affected by supplies of byproduct feed from biofuel manufacture. In this chapter, I present a life cycle assessment (LCA) of an integrated system producing animal and biofuel products. As suggested by Dale [576], I focus here on greenhouse gas (GHG) impact, and not energy balance. The fact that regulations stipulate fuels on a GHG basis (and not a petroleum energy basis) supports the focus on GHGs. First, an overview of biofuel policy is presented and important aspects of biofuel production are reviewed. Next, an integrated biorefinery and animal production system is presented and an LCA is performed to assess the potential for reducing the GWP impact of biofuels and animal production. I conclude with a thought experiment regarding the impacts of biofuel policy on animal farming and associated changes in GWP.

6.2 Liquid biofuel production and policy

The Energy Independence and Security Act of 2007 requires increasing amounts of liquid biofuel to be blended with gasoline and diesel, reaching a total of 136 gigaliters (GL; one L= 0.24 gal) annually in yr2022. The act specifies three classes of biofuel: Renewable fuel, Advanced biofuel, and Cellulosic biofuel, and these classes are defined to have life cycle GWP of 20%, 50% and 60% less than baseline gasoline, respectively. The requirement for the Renewable fuel (i.e. corn ethanol) reaches 57 GL in yr2015, and is constant through 2022. By yr2022, 80 GL of Advanced biofuel are to be produced, of which 61 GL are to be sourced from cellulosic feedstock [577] (**Figure 6-1**). The US EPA's regulatory impact analysis assumes that 57 GL biofuel will be produced from corn grain, 30 GL from dedicated non-corn energy crops, 19 GL from corn stover, 0.4 from forestry biomass, and 9 GL from urban waste (from [578], tables 1.1-23 and 1.2-3).

On an energy basis, 136 GL of ethanol (EtOH) amounts to 3.2 EJ (10^{18} J) [579]. In yr2005, 31 EJ of fuel was used in the transportation sector in the US, of which 23 EJ was used in highway vehicles (cars, light trucks, buses, motorcycles, and medium and heavy trucks). Thus, the biofuel requirement in 2022 amounts to 14% of the energy used for highway (on-road) transport in yr2005. Another way to gauge the magnitude of the EtOH mandate is to compare required production levels with recent use of all liquid fuels in the transport sector (gasoline, diesel, and

EtOH). If liquid fuel use in the transportation sector declines to 20 EJ in yr2022, the EtOH mandate would be 16% of liquid fuels used in transportation (HHV energy basis, **Figure 6-1**).

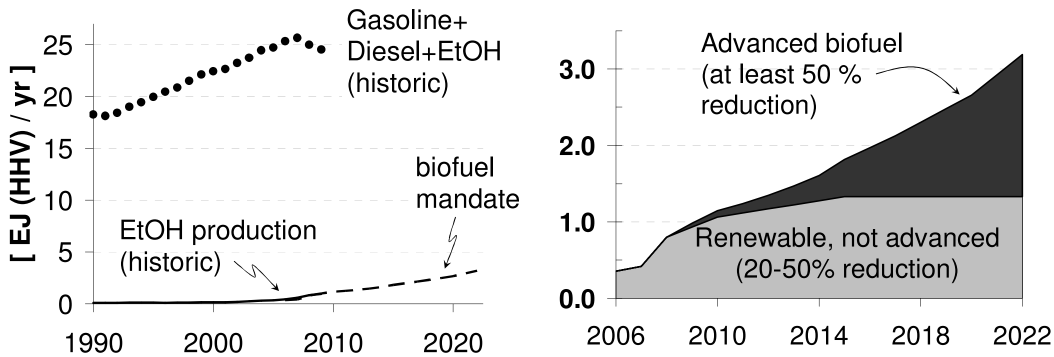

Figure 6-1. Left panel: Ethanol (EtOH) production (solid line) [580] and biofuel blending requirements (broken line) [577]. Dots show the combined use of gasoline, diesel and EtOH used in the transport sector in the US [581, 582]. Right panel: Renewable fuel requirements in the US. Units are exajoule (EJ, HHV basis) per year.

EtOH production in the US has increased from 0.14 EJ in yr2000 to over 0.90 EJ in yr2009 [580]. Brazil produced 0.62 EJ of EtOH in the 2008/2009 marketing year [583]. Thus, while the share of EtOH in liquid fuel is higher in Brazil, the US is now the largest producer of fuel EtOH in the world.

The European Union also has a biofuels mandate, which requires that 10% of liquid fuel will be bio-based in 2020, with the stipulations that the biofuel is "sustainable", and that second-generation biofuels become available. The legislation proposal notes that "The full carbon effects of such [land] conversion should therefore be accounted for in calculating the greenhouse gas savings of particular biofuels and other bioliquids." [584] (article 37).

In the US, yield increases could continue to supply more biomass without utilizing more land. Corn grain yields have increased 0.12 t/ ha annually since yr1950 (**Figure 6-2**). In order to supply corn grain in yr 2022 at the current per capita level of 0.75 t dry matter (DM)/ p/ yr (~35 bu/ p/ yr), projected population growth [585] will require an additional 33 Mt DM of corn grain per year (1.5 G bu/ yr), up from 282 Mt DM in yr2008. Assuming continued linear corn yield increases, this status-quo per capita corn supply in yr2022 could be supplied with 31.6 M ha, 0.6 M ha fewer than under corn in yr2008.

The requirement for cellulosic biofuel in the US Renewable Fuel Standard indicates the desire of legislators and policy-makers to avoid continued growth of grain use for fuel production. If cellulosic EtOH is not available (either due to conversion technology shortcomings or feedstock supply concerns), the EPA can reduce production requirements by issuing "Cellulosic biofuel waiver credits" [577]. The scale of agricultural production necessary to supply the mandated biofuel volumes suggests a substantial influence on the biomass appropriation systems in the US.

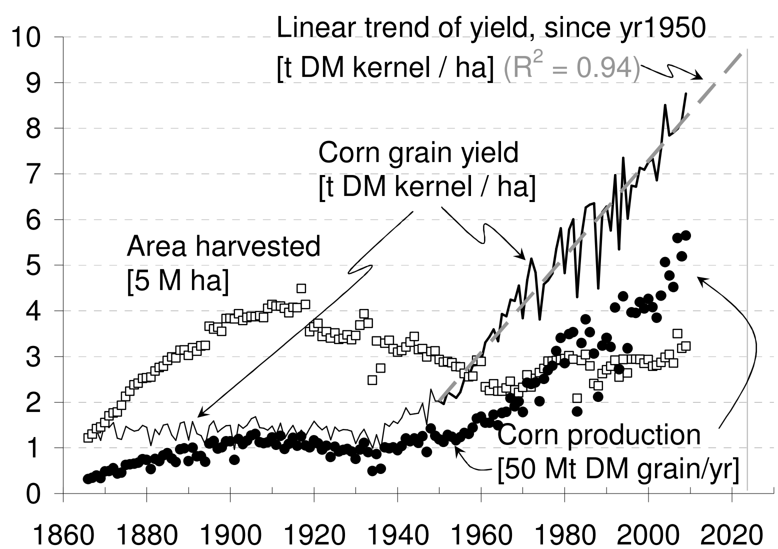

Figure 6-2. US average corn grain yield (solid line), production (dots), and area harvested (squares) [43]; note data series labels for units. Area harvested for corn grain peaked in the years around 1915. Corn grain yield has increased, with pronounced fluctuations, at about 0.12 t/ yr since yr1950. If yield growth were to continue growing linearly, the US average corn grain yield in yr2022 would be 11.6 t/ ha (185 bu/ acre).

6.3 Background on ethanol & bioenergy studies

Recent LCAs find that starch-based EtOH reduces GHG emissions compared with petroleum-based gasoline on an energy basis [120, 159-162, 586]. Such studies have led to the generally accepted net GHG impact of 55-65 g CO_2eq/ MJ for corn-based EtOH, compared to 90-95 for gasoline [587]. This net climate impact does include a credit for co-product feed, but does not include any emissions from land use change (LUC).

Co-products of biofuel production (whether food, feed, or thermal and electrical energy) are important when considering life cycle emissions. The fodder co-product (often called distillers' grains) of corn-based ethanol is often assumed to offset the need for another feed [374]. The emission credit associated with co-product feed is 15-20 g CO_2eq/ MJ corn EtOH. Drying these distillers grains is usually necessary to allow storage for use as animal feed. Co-locating animal growing operation with EtOH facilities enables feeding cattle wet byproduct and reduced emissions associated with drying [162, 588] (see **§6.4** for analysis).

Estimates of market-mediated emissions due to LUC range from 13 g CO_2 per MJ (HHV) [589] to 24 g CO_2 per MJ [196]. One of the first studies that used an economic model to calculate land use impacts reported much higher emissions [203], but it did not account for barriers to biomass trade [196]. A recent assessment by Plevin et al. finds considerable uncertainty in estimates of LUC emissions, and that the estimates above are at the lower end of plausible ranges, indicating the need for the biofuel policy to explicitly address LUC issues and uncertainty [201].

The impacts of EtOH are often compared to the impacts of gasoline on an equivalent energy basis. However, a biofuel requirement may reduce price and demand for petroleum, leading to a rebound in petroleum consumption. Thus, one MJ of EtOH may offset less than one MJ of gasoline, which would tend to reduce the effectiveness of biofuel requirements as GHG a mitigation policy [590].

EtOH from plant material rich with lignin and cellulose tend to have lower life cycle GHG emissions than corn-based EtOH [144, 591-593]. Part of the calculated improvement over corn EtOH results from lower energy and fertilizer requirements for producing the grassy and woody plants. In addition, plant material not converted to liquid fuel (e.g. recalcitrant lignin) is burned in co-generation systems to provide thermal energy for the EtOH production processes, and perhaps electricity for export to the grid. Thus, while using fewer inputs and having correspondingly lower GHG emissions during the agricultural phase, cellulosic systems also avoid some of the natural gas, coal and grid electricity required for EtOH manufacture, and may receive a GHG credit for electricity produced. Soil under perennial grasses contains more C than under annual crops [594], which may further reduce impact.

Biorefineries can produce commodity products like fuel and feed as well as higher-value products for industrial use (e.g.[595]). The types of co-products produced in biorefineries are feedstock- and process-dependent. For example, herbaceous crops can be produced that contain protein (up to 10% mass fraction) in addition to (hemi)cellulose and lignin. Thus, land under herbaceous energy crops can do “double duty”, by providing both energy feedstocks and protein for feed [164]. Any technical or economic analysis should consider the impacts of co-products on biomass markets [596].

Most EtOH in the US is produced by fermenting corn starch, and collecting the leftover corn matter in distillers grains and solubles (DGS). Facilities which ferment the whole corn kernel are often called "dry mill", although some fractionate the corn kernel before distillation. The DGS co-product of corn dry-milling is fed to animals. For every kg of corn DM that enters a dry-mill EtOH plant, about 0.38 kg DM of DGS are produced [162, 311].

The production magnitude of EtOH co-product feed is substantial and has recently surpassed the amount of soymeal fodder use in the US. In yr2009, EtOH production volume was 10.6 Ggal, with co-product distillers grains and solubles production at about 33 M tonne DM. For reference, the mass of corn grain and soybean meal fed to livestock in the US in yr2008 was 115 and 26 Mt DM, respectively. The renewable fuels standard calls for 21 Ggal of EtOH, not including the cellulosic requirement. This large supply of EtOH co-product feed is influencing the economics of feeding animals [597-599].

6.4 Impacts of an integrated biorefinery and animal farm

Incorporating animal production in the LCA of corn EtOH allows a more complete analysis of impacts related to biofuels [600]. In this section, such an integrated assessment is presented. The LCAs of animal products (**Chapter 3**) and of manure to power (**Chapter 5**) presented earlier are combined with LCA studies of EtOH from the literature described above (**§6.3**) to assess the GHG impacts of a combined biofuel and animal product system. The economics of a similar system have been previously investigated by Braden et al. [601], and an LCA conducted by Liska et al. included energy harvesting from manure [162]. The work presented here builds on this previous research by including all emissions from animal production within the system boundary and by considering a mix of animals, and not only beef cattle.

The integrated system is outlined in **Figure 6-3**. All input and output flows are characterized per second, or in terms of the mass of corn grain dry matter (DM) entering the biorefinery. The size of the system is defined by the EtOH production rate, and all EtOH byproducts are consumed by animals as feed.

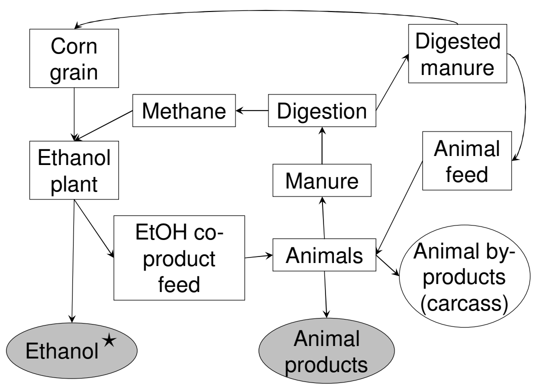

Figure 6-3. Conceptual diagram of the integrated biorefinery and animal farm. The ★ on Ethanol indicates that the ethanol production rate defines the size of the system.

6.4.1 The EtOH production sub-system

The EtOH sub-system produces 4.8 liters of anhydrous ethanol per second (150 ML/ yr). Starch and sugar-based EtOH production are mature bio-chemical refining processes. Corn grain yields about 0.48 L EtOH (anhydrous, not denatured) per kg DM grain input [311]. Thus, corn grain enters the EtOH subsystem at a rate of 10 kg DM per second.

The energy inputs and resulting GHG emissions associated with EtOH manufacture are shown in **Table 6-1**. The life cycle GHG impact of electricity used at an EtOH plant is assumed to be 180 g CO_2eq/ MJe (650 g CO_2eq/ kWh), and the life cycle GHG impact of natural gas is 60 g CO_2eq/ MJ (HHV) [123]. The electricity requirement is 3.1 MWe, while the heat requirement during EtOH production, not including any co-product drying, is 25 MW thermal (MWth). Feeding co-product to livestock as a wet or moist feed can reduce natural gas requirement at the EtOH plant by avoiding drying. If the co-product feed is to be shipped more than a few tens of km, drying is necessary [588].

Table 6-1. Electricity and heat (as natural gas) requirements during corn grain EtOH manufacture, and associated GHG emissions. Energy inputs are shown per liter of EtOH, per MJ of EtOH, and per kg corn DM input [161, 162, 311].

		Electricity	Natural gas: co-product drying	Natural gas: all other
Energy inputs	[MJ/ L EtOH (anhydrous)]	0.65	3.41	5.29
	[MJ/ MJ EtOH (HHV)]	0.03	0.15	0.23
	[MJ/ kg DM corn input]	0.31	1.64	2.54
GHG emission	[g CO_2eq/ kg DM corn input]	60	98	152

In addition to the energy used on-site, emissions associated with building and maintaining the EtOH plant amount to 4.7 g CO_2eq/ kg DM corn input and emissions from transporting corn to the EtOH plant are 23 g CO_2eq/ kg DM corn input [161, 162].

For each kg of corn grain DM that enters the EtOH plant, between 0.35 and 0.42 kg DM of co-product feed are produced [162, 311]. For this assessment, the average of these two estimates is used: 0.38 kg DM of co-product feed is produced for each kg of corn DM entering the plant. The co-product feed, also called distillers grains, includes any part of the corn grain that was not converted to EtOH.

In the integrated scenario, the animal production and EtOH facility are co-located, so transportation is assumed not to limit the feeding of wet DGS. However, DGS fed to poultry are assumed to be dried.

6.4.2 Animal production

The animal production subsystem consists of a mixed animal farm producing various animal products. In practice, animal farms are specialized, but the hypothetical mixed farm is adopted in order to account for the likelihood that EtOH co-product feed (DGS) will be consumed by all animal types [580]. DGS are assumed to replace non-pasture mixed fodder rations. This assertion is supported by Beckckman et al. who find that DGS prices are more closely tied to energy feed prices (e.g. corn) than to prices of protein feed (e.g. soymeal) [602]. The DGS is assumed to replace 15% of non-pasture feed for all animal types, except beef cattle on feed, for which DGS replace 20% of feed [603]. DGS are produced by the EtOH plant at a rate of 3.8 kg DM/ second.

The composition of animals consuming DGS is assigned such that the ratios of animal product (AP) outputs are the same as national AP production ratios in yr2008 (fraction of total kcal produced supplied as a given AP). The DGS fed to poultry is assumed to be dried, while the DGS fed to pigs and cattle is wet. The baseline feed consumption for animals not eating DGS, along with the GHG impact of the feed mixes, are shown in **Table 6-2 (a)**. The performance of beef cattle consuming DGS is improved such that the cattle gain weight 6% faster than they would without the DGS [604]. As a result, the cattle on feed consuming DGS consume less feed and subsequently emit less enteric methane (CH_4). The performance of other animals eating DGS is unchanged. Feed consumption for the animals on the integrated farm is shown in **Table 6-2 (b)**.

Table 6-2. Feed consumption during animal production. The baseline feed consumption and GHG impact refer to animals not consuming DGS, as modeled in Chapter 3.

(a) baseline		Dairy	Chicken (meat)	Beef (on feedlot)	Beef (not on feedlot)	Pig	Egg	Turkey
Non-pasture feed input	[kg DM / 200 kcal serving]	0.29	0.35	0.40	0.55	0.58	0.50	0.44
Pasture feed input	[kg DM / 200 kcal serving]	0.02	0.00	0.00	1.22	0.00	0.00	0.00
GHG impact of non-pasture feed	[g CO2eq / kg DM]	248	402	318	232	402	402	413
(b) with DGS feeding								
DGS consumed	[fraction of DGS]	0.32	0.13	0.25	0.00	0.21	0.05	0.03
DGS feed	[kg DM / 200 kcal serving]	0.043	0.052	0.074	0.000	0.087	0.075	0.065
Non-DGS, non-pasture feed	[kg DM / 200 kcal serving]	0.245	0.295	0.297	0.549	0.493	0.424	0.371
Pasture	[kg DM / 200 kcal serving]	0.02	0.00	0.00	1.22	0.00	0.00	0.00

6.4.3 Manure management subsystem

In the integrated system, energy is harvested from manure produced by animals, and is used to provide heat or electricity. The manure production rates are from **Chapter 5**, except for beef cattle, which is pro-rated to account for the lower total feed intake. Manure from cattle, hogs and egg-producing hens is assumed to be digested anaerobically to produce biogas; manure from other poultry is assumed to be dryer and is combusted. Manure energy yields from each animal type are shown in **Table 6-3**. The impact of manure-sourced electricity is estimated at 20 g CO_2eq per MJ electricity (MJe), while the GHG impact of the biogas used as heat is 10 g CO_2eq per MJ thermal (MJth, HHV). See **Chapter 5** for details of the energy harvesting LCA.

Table 6-3. Energy yield and production rate of manure. Energy yield is expressed per second, and per kg of manure dry matter excreted by the animal (kg DMx). For chickens (meat) and turkey manure is a solid fuel; for other animals the manure energy harvesting is carried out with anaerobic digestion, yielding biogas.

		Dairy	Chicken (meat)	Beef (on feedlot)	Pig	Egg	Turkey
Manure energy production	[MJ (HHV)/ s]	15.6	15.5	4.0	5.9	3.2	2.8
Manure energy yield	[MJ (HHV)/ kg DMx]	4.6	12.8	3.6	7.4	13.1	14.0

6.4.4 Results

For both the baseline and integrated system scenarios, EtOH is produced at a rate of 4.8 liter per second, and animal food is produced at a rate of 65 servings (200 kcal) per second. The net GHG impacts of the baseline EtOH system amount to 6.3 kg CO_2eq per second (this estimate uses the average of GREET and BESS estimates presented by Plevin [161], which include an emission credit for DGS), while the impact of the baseline animal production system is 25 kg CO_2eq per second.

For the integrated system, energy harvested from animal manure is used to power the EtOH facility. With 30% (HHV) efficiency of electricity conversion, the gross manure energy required to supply necessary electricity and heat (not including DGS drying) is 35 MWth. The drying of co-product feed (consumed by poultry) requires 4 MWth. The balance of available manure energy is 7.5 MWth. With a capacity factor of 0.85, the electricity exported amounts to 0.5 kWh/ s (1.9 MWe). Biomass enters the integrated system at a rate of 30 kg DM/ s, of which 80% is consumed by animals.

Due to the construction and maintenance of MtoP infrastructure, the manure-sourced heat and electricity is not fully climate-neutral, but the GHG impact is substantially lower than grid electricity and fossil natural gas. The GHG impact of the energy supplied to the EtOH via manure energy harvesting is 0.4 kg CO_2eq per second, compared to 3.1 kg CO_2eq per second from the baseline EtOH plant. The electricity not required at the EtOH plant generates GHG credits at a rate of 0.35 kg CO_2eq per second.

Finally, we can compare the GHG impact of the two baseline systems with the integrated system. The baseline system emits a total of 32 kg CO_2eq per second, whereas the integrated system emits 23 kg CO_2eq per second – a reduction of 28%. The majority of the emission reduction (70%) comes from the animal subsystem. Emissions during animal production are reduced in the integrated system primarily by lower CH_4 emissions during manure management, but also by using the feed co-product from the EtOH facility (**Figure 6-4**).

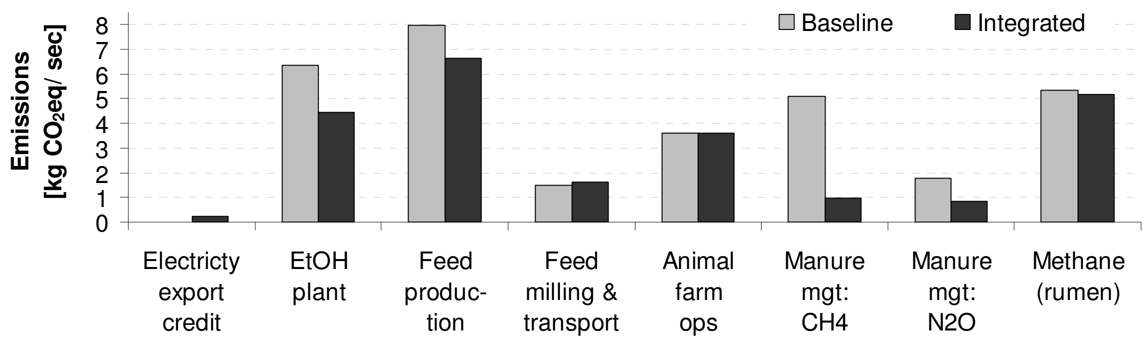

Figure 6-4. GHG emissions from the baseline (light bars) and integrated (dark bars) systems. Emissions reductions from the animal subsystem make up 70% of total emission reduction. EtOH plant emissions are reduced through low-C energy harvested from manure.

With the use of energy harvested from the manure, the emissions associated with the EtOH subsystem were 4.5 kg CO_2eq per second – a decrease of 38% from the baseline. Emissions associated with the EtOH production subsystem correspond with life cycle emissions of 40 g CO_2eq per MJ EtOH (not including any impacts from land use change). If, however, all of the emissions reductions from the integrated system are assigned to the EtOH, the fuel is produced with negative emissions (-18 g CO_2eq/ MJ).

The achievement of such a carbon-negative fuel depends on the specifics of GHG emission policies. If agriculture is allowed to sell emission credits in a regulated market (or if it is offered revenue streams for reducing emissions), then the fuel could be considered to carry such low GHG emissions. In practice, animal producers have been offered grants and subsidies to install manure treatment equipment, and under this circumstance, the emission reductions from manure management should probably be associated with the subsidy, and not the integrated system. Even if the EtOH is not credited with the emissions reductions from animal farming, the low-C energy harvested from manure does reduce impacts of EtOH production. With such a system, corn EtOH offers 50% GHG reduction compared to gasoline, and thus may be considered an advanced biofuel, as defined and required in the Renewable Fuels Standard.

6.5 Thought experiment: shifting animal production

If biofuel policies that require non-grain feedstock increase demand for cellulosic biomass, the price of roughages used to feed cattle will likely be affected. Cellulosic-feedstock-based biofuel production is sought after partly because such feedstocks do not compete directly with grain. The cattle industry is a large consumer of hay and grazed cellulosic biomass, while poultry and hog production uses mostly concentrate feeds (see **Chapter 3**). In addition, as cattle and biorefineries compete for resources, formerly grazing cattle may be fed increasing amounts of co-product feeds. Thus, increasing cellulosic biomass requirements for EtOH will affect cattle production disproportionately compared with other animals.

Fodder use could be shifted by adopting alternative animal production strategies (particularly for beef), and by changes in meat consumption patterns. These changes would be associated with lower life cycle global warming potential (GWP), water, and land conversion impacts. In this thought experiment, I first assume that the cellulosic biofuel requirement will cause ¼ of the beef protein supply (currently 1.33 Tg/ yr, see **Chapter 3**) to be supplied instead by chicken meat, which is less roughage-dependent than beef. Using the work presented in **Chapter 3**, chicken meat is associated with 11 g CO_2eq/ g protein, while beef is associated with 108 g CO_2eq/ g protein. Thus, each gram of protein supply shifted from beef to chicken saves 97 g CO_2eq. The cellulosic biofuel requirement is 1.4 EJ/ yr. Dividing this annual cellulosic biofuel requirement by the annual GHG savings associated with reduced beef and increased chicken production results in a GHG credit of about 23 g CO_2eq/ MJ of fuel. This credit is similar to the feed co-product credit routinely granted corn grain EtOH, and illustrates the potential for cellulosic feedstock-promoting policies to have non-uniform impacts on animal feeding.

7 Summary

Through land use and biomass utilization, humans are dominant forces in the planetary biosphere and carbon (C) and nitrogen cycles. In this dissertation, two major components of human biomass appropriation in the United States (US) are studied: animal feeding and energy service provision.

In Chapter 1, I review biomass use by humans and the greenhouse gas (GHG) implications of using biomass to supply energy services. Human appropriation of global terrestrial net primary productivity (HANPP) is a ratio of appropriation and production; it currently stands at 15-30% of NPP. Over half of used biomass is fed to animals, while equal amounts are eaten directly by humans and burned for energy services. I show how disproportionate per capita meat consumption across countries indicates the potential for increased animal feed demand. I then present a life cycle assessment (LCA) of growing corn and switchgrass in the US, incorporating uncertainty in key impact categories. With respect to climate change, uncertainty regarding soil carbon storage capacity, nitrous oxide (N_2O) emissions, and emissions from land use change (LUC) influence the attractiveness of bioenergy (bioE) for GHG abatement. A discussion of the linkages between HANPP, GHG emissions, and land use change follows. The "baseline" situation regarding the C stock in terrestrial ecosystems is a critical factor in assigning responsibility for changes in ecosystem C storage. Biomass policy could also influence crop trade and animal production, so I conclude Chapter 1 with an estimate that appropriating all US biomass exports and all roughage fed to beef cattle would supply about 8 EJ of bioE feedstock.

Improving biomass yield and expanding harvested area can increase supply of bioE feedstock, but so too can more efficient resource utilization. In Chapter 2, I review estimates of the bioE feedstock potential for byproduct biomass in the US. The magnitude of the organic byproduct resource is up to 11 EJ, when large amounts of crop residue are considered. In some cases, the efficiency improvement will appropriate a biomass flow that would have otherwise been used by "nature" (e.g. C from crop residues that is plowed into fields respired as CO_2), while in other cases, biomass byproduct use will reduce some other impacts (e.g. collecting CH_4 emissions from landfills). In order to put the GHG benefit of various byproduct management practices in perspective, I also propose a simple life cycle framework, and apply it to municipally sourced organic byproduct management. If CH_4 emissions from landfills are uncontrolled, net GWP is high. But if CH_4 collection efficiency is around 70%, landfills, anaerobic digestion and composting all have comparable impacts. Thus, the cost associated with depositing material in landfills and the revenue associated with composting will drive management decisions.

There have been many studies of the life cycle global warming potential (GWP) impact of animal products, but I found a dearth of research that synthesizes the biomass demand and GWP of the major animal products produced in the US in a differentiated but inclusive way. Thus, I offer an LCA of animal feeding in the US, which includes dairy (milk), beef, chicken (meat), pig (meat), egg, and turkey (meat) production. Land requirements, irrigation water use, and GWP are estimated for each animal product (AP). I show that the impact of producing animals (at the farm gate) in the US accounts for about 5% of total GWP. The LCA reveals that harvested roughage (e.g. grass, alfalfa, silage), grazed biomass, and corn grain each supplies roughly 30% of all feed use in the US (dry matter mass basis). In addition, only 10% of methane (CH_4) emissions related to beef production occur on feedlots (23% of total GWP from beef is attributable to feedlots).

These results highlight the US cattle sector as a large consumer of non-grain feeds, and the potential for increased cellulosic biomass utilization as bioE to influence cattle markets.

Animal production in the US is dominated by large concentrated animal feeding operations (CAFOs), and manure management has emerged as a major issue for CAFO managers. In Chapter 4, I first calculate the potential economic value of animal manure, including nutrients (NPK), energy, and fiber. I then estimate the cost of producing biogas from CAFO manure, and compare this cost with a range of operational revenues. Biogas production generally costs \$2-\$4 per GJ (HHV), but gas conditioning can add variable cost, depending on the H_2S and moisture tolerances of gas use equipment. Removing carbon dioxide (CO_2) to produce high-purity CH_4 adds roughly \$4 per GJ. Finally, in Chapter 4 I show that the cost of treating manure with anaerobic digestion adds 0.5%-1.5% to the current cost of animal production.

An LCA of harvesting energy from manure is performed in Chapter 5. The impact of biogas is about 10 g CO_2eq/ MJ, considering only required process energy and equipment. In addition to these process-related emissions, consequential emissions (or reductions) may occur from fugitive CH_4, or from eliminating manure storage CH_4 emissions. The GHG impact of manure-to-power schemes (MtoP) is compared to the impacts of animal production, and I show that harvesting energy from manure can reduce or offset 3% - 45% of the AP life cycle emissions, depending on whether an MtoP project is responsible for emissions reductions associated with manure. In these circumstances, MtoP is shown to be a cost-effective GHG mitigation strategy. For pig, dairy and egg production, manure management emission reductions account for more than half of reductions; for the other AP systems, the energy credit dominates any reduction.

In Chapter 6, the results from the AP LCA and the MtoP LCA are applied to perform an LCA of an integrated AP and bioE facility. Biomass flows and GWP are modeled for two systems: one where the AP and bioE facilities are distinct and one where the facilities are integrated. The two systems supply the same outputs: a mix of animal products and liquid biofuel. Overall, GHG emissions are 28% lower from the integrated facility. The animal product sub-system consumes 80% of the biomass input, and accounts for 70% of the overall GWP reductions. A liquid fuel with negative GHG impact can be produced if it is credited with all GWP reduction associated with the integrated system, although such a negative GWP depends on the determination of a "baseline" emission from animal production with substantial emissions from manure. I conclude with a hypothetical whereby cellulosic biofuel policies have differential effects on cattle production and cause animal production to shift away from beef to less GHG-intensive meat.

8 References

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