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USDA Agriculture and Forestry Greenhouse Gas Inventory: 1990–2008

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

Abstract

Emissions of the three most important long-lived greenhouse gases (GHG) have increased measurably over the past two centuries. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) concentrations in the atmosphere have increased by approximately 38%, 143%, and 18%, respectively, since 1750. In the U.S., agriculture accounted for approximately 6% of total GHG emissions (6,957 Tg CO_2 eq. [teragrams of carbon dioxide equivalent]) in 2008. Livestock, grasslands, crop production, and energy use contributed a total of 462 Tg CO_2 eq. to the atmosphere in 2008. This total includes an offset from agricultural soil carbon sequestration of roughly 40 Tg CO_2 eq. The primary agricultural sources are N_2O emissions from cropped and grazed soils (214 Tg CO_2 eq.), CH_4 emissions from enteric fermentation (141 Tg CO_2 eq.), CO_2 emissions from energy use (72 Tg CO_2 eq.), and CH_4 emissions from managed livestock waste (45 Tg CO_2 eq.). Forests in the United States contributed a net reduction in atmospheric GHG of approximately 886 Tg CO_2 eq. in 2008, which offset total U.S. GHG emissions by approximately 13%. In aggregate, the U.S. agricultural sector (including GHG sources from crop and livestock production, grasslands, energy use and GHG sinks for forests and urban trees) was estimated to be a net sink of 424 Tg CO_2 eq. in 2008.

Keywords: climate change, greenhouse gas, land use, carbon stocks, carbon sequestration, enteric fermentation, livestock waste, nitrous oxide, methane, rice cultivation, energy consumption.

June, 2011

Dear Reader:

I am pleased to present you with this report, *The U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2008*, an update to USDA Technical Bulletin 1921 (2008), which accounted for greenhouse gas emissions and sinks for the agricultural and forestry sectors through 2005.

This report is consistent with the U.S. Environmental Protection Agency's (EPA) *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (April, 2010) in its assessment methods. However, EPA's national-scale reporting here has been disaggregated to provide a State-by-State presentation. We believe this format will serve as a useful resource to land managers, planners, and others with an interest in greenhouse gas dynamics and their relationships to land use and land use change.

Data collection and analysis, as well as coordination of this *Inventory,* could not have been accomplished without the contributions of Stephen Del Grosso, Ronald Follett, and others within USDA's Agricultural Research Service. I also express my thanks to Linda Heath, James Smith, and Rich Birdsey of the USDA Forest Service; James Duffield of USDA's Office of Energy Policy and New Uses; Jerry Hatfield of USDA's Agricultural Research Service; Stephen Ogle at the Natural Resources Ecology Laboratory of Colorado State University; and Tom Wirth in EPA's Office of Atmospheric Programs for their data, analysis, and review. Their thoughtful and diligent efforts compose the foundation of this report, which we hope will serve as a useful resource for a broad spectrum of land management-focused professionals and other interested individuals.

Sincerely,

William Hohenstein
Director, USDA Climate Change Program Office

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This report was made possible by contributions from a number of individuals and collaboration between the U.S. Department of Agriculture (USDA), the U.S. Environmental Protection Agency (EPA), and Colorado State University.

The U.S. Agriculture and Forestry Greenhouse Gas Inventory (USDA GHG Inventory) is supplemental to the official Inventory of U.S. Greenhouse Gas Emissions and Sinks (U.S. GHG Inventory) submitted by EPA to the United Nations Framework Convention on Climate Change each April. We thank the EPA for permission to reprint estimates and methodologies from the official U.S. GHG Inventory. We would like to acknowledge the contribution of Tom Wirth of EPA's Office of Atmospheric Programs, who provided detailed emissions data for livestock sources of methane and nitrous oxide reported in Chapter 2. Stephen Del Grosso, Melissa Reyes-Fox, and Kris Nichols of USDA Agriculture Research Service (ARS) Northern Plains Area (NPA) Soil Plant Nutrient Research unit in Fort Collins, CO, in collaboration with Stephen Ogle, William Parton, and Cindy Keough of Colorado State University, provided estimates for N_2O emissions from agricultural soils, wrote most of the text for Chapters 2 and 3, and all of the text for Chapter 1. The Natural Resources Ecology Laboratory (NREL) of Colorado State University prepared the agricultural soil carbon estimates and supporting text for Chapters 2 and 3. Amy Swan of the NREL assisted with maps for Chapter 3. James Smith and Linda Heath of the USDA Forest Service, Northern Research Station, provided estimates and text for Chapter 4. The USDA Forest Service tailored its analysis for this report to partition regional and land ownership trends in forest carbon. James Duffield of the Office of Energy Policy and New Uses prepared Chapter 5. The estimates presented in Chapter 5 are unique to this report and derive from ongoing work in the Office of Energy Policy and New Uses to track fuel consumption in agriculture. We acknowledge Mary F. Smith and Morgan Carey of USDA ARS NPA Soil Plant Nutrient unit for formatting the document for printing.

Comments provided by reviewers from the USDA ARS, EPA, USDA Forest Service, and Colorado State University greatly improved this document. Brenda Chapin, Office of the Chief Economist, and the USDA Office of Communications provided assistance with publishing.

Glossary of Terms and Units

CO_2	Carbon dioxide
CH_4	Methane
N_2O	Nitrous oxide
NOx	Nitrogen oxides
C	Carbon
GHG	Greenhouse gas
GWP	Global warming potential
Tg	Teragram (10^{12} grams)
Tg CO_2 eq.	Teragrams of carbon dioxide equivalent
Gg	Gigagram (10^{9} grams)
Mg	Megagram (10^{6} grams)
t	Metric ton (1,000 kg)
ha	Hectares
DE	Digestible energy (percent)
Y_m	Fraction of gross energy converted to CH_4
TDN	Total digestible nutrients
VOCs	Volatile organic compounds
VS	Volatile solids
B_o	Maximum CH_4-producing capacity for domestic wastewater
DM	Dry matter
Btu	British thermal unit
Qbtu	Quadrillion British thermal units
Tbtu	Trillion British thermal units
EF	Emission factor
MCF	Methane conversion factor

Chapter 1: Introduction

1.1 Global Change and Global Greenhouse Gas Emissions in Agriculture and Forestry

In 2008, total U.S. greenhouse gas emissions measured 6,957 teragrams of carbon dioxide equivalents (Tg CO_2 eq.), rising nearly 14 percent from 1990 estimates (EPA 2010). Global concentrations of the three most important long-lived greenhouse gases (GHGs) in the atmosphere have increased measurably since the onset of the Industrial Revolution in 1750. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) concentrations in the atmosphere have increased by approximately 36%, 148%, and 18% respectively (EPA 2010, Keeling & Whorf 2005, Dlugokencky et al. 2005, Prinn et al. 2000). Agriculture and forestry practices may either contribute to or remove GHGs from the atmosphere. Agriculture and forestry have contributed to GHG levels in the atmosphere through cultivation and fertilization of soils, production of ruminant livestock, management of livestock manure, land use conversions, and fuel consumption. The primary GHG sources for agriculture are N_2O emissions from cropped and grazed soils, CH_4 emissions from ruminant livestock production and rice cultivation, and CH_4 and N_2O emissions from managed livestock waste. The management of cropped, grazed, and forestland has helped offset GHG emissions by promoting the biological uptake of CO_2 through the incorporation of carbon into biomass, wood products, and soils, yielding a total U.S. net emissions of 6,016 Tg CO_2 eq. (net CO_2 flux from Land Use, Land Use Change, and Forestry, EPA 2010). This report serves to estimate U.S. GHG emissions for the agricultural and forestry sectors, to quantify uncertainty in emission estimates, and to estimate the potential of agriculture to mitigate U.S. GHG emissions.

Table 1-1 Agriculture and Forestry Greenhouse Gas Emission Estimates and Uncertainty Intervals, 2008

	Estimate	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Source		*Tg CO_2 eq.*[1]			*percent*
Livestock	203	185	230	-9	+14
Crops[2]	154	84	215	-34	+71
Grassland[2]	33	5	132	-84	+298
Energy Use[3]	72				
Forestry	(792)	(935)	(651)	-18	+18
Urban Trees	(94)				
Net Emissions	**(424)**	**(587)**	**(240)**	**-38**	**+44**

Note: Parentheses indicate net sequestration.
[1] Teragrams of carbon dioxide equivalent.
[2] Includes sequestration in agricultural soils.
[3] Confidence intervals were not available for this component.

Observed increases in atmospheric GHG concentrations are primarily a result of fossil fuel combustion for power generation, transportation, and construction. In the United States, agriculture accounted for 6.1% of total GHG emissions in 2008 (EPA 2010). Greenhouse gas emissions estimates reported here are in units of CO_2 equivalents. Box 1-1 describes this reporting convention, which normalizes all GHG emissions to CO_2 equivalents using Global Warming Potentials (GWP). Agriculture in the United States, including livestock, grasslands, crop production, and energy use, contributed a total of 462 Tg CO_2 eq. to the atmosphere in 2008 (Table 1-1). This total includes an offset, or sink, from agricultural (cropped and grazed lands) soil carbon sequestration of roughly 40 Tg CO_2 eq (Table 1-2). Forests in the United States contributed a net reduction in atmospheric GHGs of approximately 886 Tg CO_2 eq. in 2008, which offset total U.S. GHG emissions by almost 13% (EPA 2010). After accounting for GHG sources and C sequestration, agricultural and forested lands in the U.S. were estimated to be a net sink of 424 Tg CO_2 eq. (Table 1-1). The 95% confidence interval for this estimate ranges from a sink of 587 to 240 Tg

BOX 1-1

The USDA greenhouse gas (GHG) Inventory report follows the international convention for reporting GHG emissions, as described in the introduction of the U.S. GHG Inventory (EPA, 2006). Emissions of GHGs are expressed in equivalent terms, normalized to carbon dioxide using Global Warming Potentials (GWPs) published by the Intergovernmental Panel on Climate Change (IPCC) Second Assessment Report (SAR). Global Warming Potentials, which are based on physical and chemical properties of gases, represent the relative effect of a given GHG on the climate, integrated over a given time period, relative to carbon dioxide (CO_2) (IPCC, 2001). The GWP values used in the U.S. GHG Inventory and this report are recommended by the IPCC for national GHG inventory reporting (Table B1-1). These values for methane (CH_4) and nitrous oxide (N_2O) are referenced to CO_2 and based on a 100-year time period (IPCC, 1996).

Table B1-1 (Reproduced from U.S. GHG Inventory 2003, Table 1-2)
Global Warming Potentials of Selected Greenhouse Gases

Gas	Atmospheric lifetime (yrs)	GWP*
CO_2	50-200	1
CH_4	12	21
N_2O	120	310

*For consistency with international reporting standards, the U.S. GHG Inventory uses GWP values published in the IPCC Second Assessment Report (1996). Global Warming Potential and estimated atmospheric lifetime values were revised for some gases in the IPCC Third Assessment Report (2001).

In the USDA and U.S.GHG Inventories, units are expressed as teragrams carbon dioxide equivalent (Tg CO_2 Eq.). One teragram equals one million metric tons. The formula for converting gigagrams (1Gg = 10^9 grams) of a GHG to teragrams (1Tg = 10^{12} grams) of carbon dioxide equivalent (Tg CO_2 eq.) is provided in the U.S. GHG Inventory and is repeated here for clarity:

$$\mathrm{TgCO_2\ eq.} = (\mathrm{Gg\ of\ gas}) * (GWP) * \left(\frac{1Tg}{1{,}000Gg}\right)$$

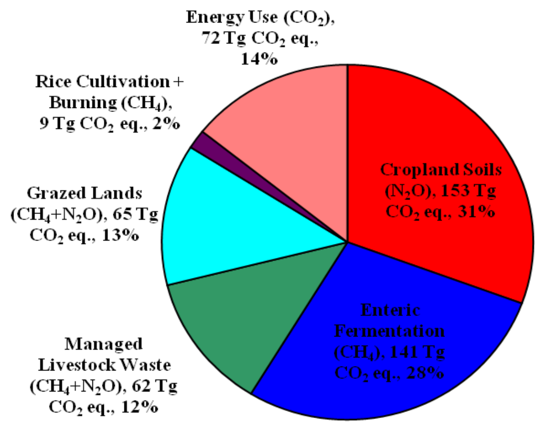
In the land use sector, where carbon dioxide gas is sequestered and stored as carbon (C) in biomass and soils, greenhouse gas removals are often expressed in units of million metric tons of carbon equivalent (MMTCE). The formula below shows how to convert MMTCE to Tg CO_2 eq., and is based on the molecular weights of carbon and carbon dioxide.

$$TgCO_2\ eq. = \ MMTCE * \left(\frac{44}{12}\right)$$

CO_2 eq. (Table 1-1). Approximately one-third of agriculture's GHG emissions in 2008 were due to crop production. Most of the emissions from crop production were from non-rice soils, with residue burning and rice cropping accounting for about 2% of overall agricultural emissions

(Figure 1-1). Livestock production is responsible for most of the remaining agricultural emissions, with about 28% from enteric fermentation, 12% from managed waste, and 13% from grazed lands. The remaining 14% of total emissions result from agriculturally related energy usage, which is listed under the Energy heading by EPA (2010), but is provided here for comprehensiveness. It should be noted that the estimates in Figure 1-1 are for emissions only and do not account for C storage in agricultural soils and forests. Regarding sequestration, forests are by far the leading sink, followed by harvested wood products, urban trees, and agricultural soils (Figure 1-2).

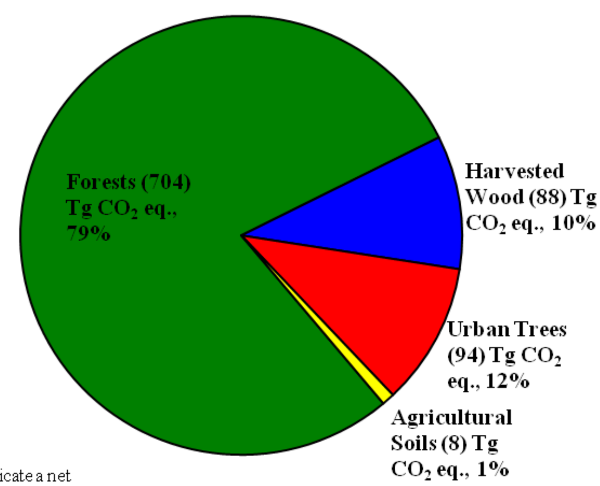
Figure 1-1
Agricultural Sources of Greenhouse Gas Emission in 2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Sources and sinks of emissions are conveniently partitioned (sinks are less than 0) in Figure 1-3. Overall emissions profiles of agricultural sources, including energy use but excluding storage by soils and forestry, show that sources increased 8% between 1990 and 2008 (Table 1-2, Figure 1-3). The sink strength of the forest pool has increased 13% since 1990 (Table 1-2, Figure 1-3). However, the sink strength of agricultural soils has decreased by approximately 57% since 1990. In aggregate, the net emissions decreased slightly from 1990 to 2008 by about 2%.

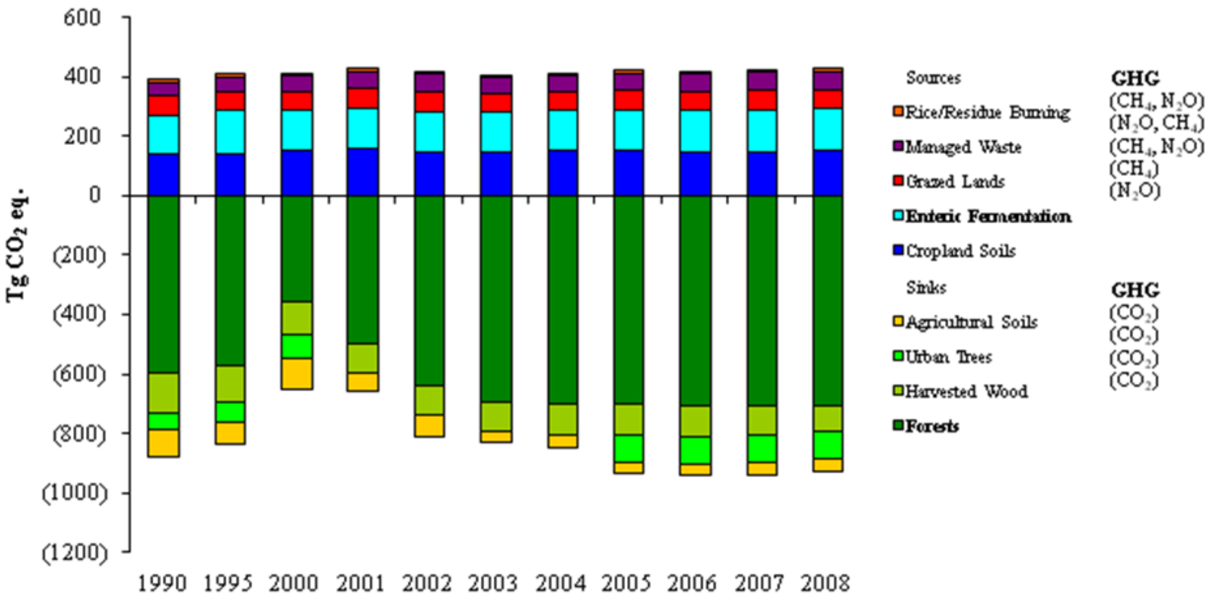
Figure 1-2
Agricultural Sinks of Carbon Dioxide in 2008



Note: Parenthesis indicate a net sequestration.
Tg CO2 eq. is teragrams carbon dioxide equivalent.

Annual CO_2 emissions from on-farm energy use in agriculture are small relative to total energy use across all sectors in the United States. In 2008, fuel and electricity consumption associated with crop and livestock operations resulted in 72 Tg CO_2 (Table 1-1), which is about 1% of overall energy-related CO_2 emissions for 2008, equaling 5,572.8 Tg CO_2 (EPA 2010). Electricity use led to about 38% of CO_2 emissions from energy use in agriculture; diesel fuel use led to about 38%, while gasoline, liquefied petroleum gas, and natural gas contributed 11%, 7%, and 5%, respectively, to total CO_2 emissions from energy use in agriculture.

Figure 1-3
Agriculture and Forestry Emissions and Offsets for 1990, 1995, 2000-2008



Note: Parenthesis indicate a net sequestration. Data not reported for CO_2 sequestration by urban trees from 2001-2004.

Tg CO_2 eq. is teragrams carbon dioxide equivalent. CH_4 is methane. N_2O is nitrous oxide. GHG is greenhouse gases.

1.2 Sources and Mechanisms for Greenhouse Gas Emissions

Over half of global annual emissions of CH_4 and roughly a third of global annual emissions of N_2O are believed to derive from human sources, mainly from agriculture (IPCC 2007). Agricultural activities contribute to these emissions in a number of ways. While losses of N_2O to the atmosphere occur naturally, the application of nitrogen to amend soil fertility increases the natural rate of emissions. The rate is amplified when more nitrogen is applied than can be used by the plants, either due to volume or timing. In agricultural practices, nitrogen is added to soils through the use of synthetic fertilizers, application of manure, cultivation of nitrogen-fixing crops/forages (e.g., legumes), and retention of crop residues. Rice cultivation involves periodic flooding of rice paddies, which promotes anaerobic decomposition of organic matter in soil from rice residue and organic fertilizers by CH_4-emitting soil microbes. Finally, burning of residues in agricultural fields produces CH_4 and N_2O as byproducts.

Livestock grazing, production, and waste cause CH_4 and N_2O emissions to the atmosphere. Ruminant livestock such as cattle, sheep, and goats emit CH_4 as a byproduct of their digestive processes (called "enteric fermentation"). Managed livestock waste can release CH_4 through the biological breakdown of organic compounds and N_2O through nitrification and denitrification of nitrogen contained in manure; the magnitude of emissions depends in large part on manure management practices and to some degree on the energy content of livestock feed. Grazed lands have enhanced N_2O emissions from nitrogen additions through manure and urine and from biological fixation of nitrogen by legumes, which are typically seeded in heavily grazed pastures. Some pastures are also amended with nitrogen fertilizers, managed manure, and sewage sludge, which also contribute to GHG emissions on those lands.

Table 1-2 Summary of Agriculture and Forestry Emissions and Offsets, 1990, 1995, 2000-2008

		1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Source	GHG	*Tg CO_2 eq.*										
Livestock		**176.1**	**193.0**	**192.0**	**192.6**	**194.3**	**189.3**	**191.1**	**195.6**	**198.6**	**204.3**	**203.0**
Enteric Fermentation	CH_4	132.4	143.7	136.8	136.0	136.3	134.5	134.6	136.7	139.0	141.2	140.8
Managed Waste	CH_4	29.3	33.9	38.6	40.1	41.2	38.4	40.2	42.2	42.3	45.9	45.0
Managed Waste	N_2O	14.4	15.5	16.7	16.5	16.8	16.3	16.4	16.6	17.3	17.3	17.1
Grassland		**(2.1)**	**5.7**	**(22.5)**	**8.9**	**(4.5)**	**30.6**	**31.6**	**33.4**	**32.1**	**31.1**	**33.2**
Grassland	CH_4	2.9	3.0	2.7	2.8	2.7	2.7	2.8	2.9	3.0	3.0	2.9
Grassland	N_2O	64.0	61.6	58.2	63.9	64.2	59.1	60.1	61.8	60.5	59.5	61.7
Grassland	CO_2	(69.0)	(58.9)	(83.4)	(57.7)	(71.4)	(31.2)	(31.2)	(31.3)	(31.3)	(31.4)	(31.4)
Crops		**124.8**	**136.9**	**136.7**	**164.5**	**156.1**	**148.4**	**152.7**	**153.6**	**148.5**	**149.2**	**153.8**
Cropland Soils[1]	N_2O	139.1	143.7	151.3	159.7	149.7	147.3	151.9	153.2	150.0	150.7	153.4
Cropland Soils[2]	CO_2	(22.6)	(15.6)	(23.5)	(4.3)	(1.7)	(7.2)	(8.3)	(8.0)	(8.9)	(9.2)	(8.3)
Rice Cultivation	CH_4	7.1	7.6	7.5	7.6	6.8	6.9	7.6	6.8	5.9	6.2	7.2
Residue Burning	CH_4	0.8	0.7	0.9	0.9	0.8	0.9	1.0	0.9	0.9	1.0	1.0
Residue Burning	N_2O	0.4	0.4	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Energy Use[3]	CO_2	**73.9**	**73.9**	**73.9**	**73.9**	**72.9**	**71.9**	**70.9**	**69.9**	**70.5**	**71.0**	**71.6**
Forestry		**(786.9)**	**(759.9)**	**(545.2)**	**(593.9)**	**(737.4)**	**(790.7)**	**(805.5)**	**(894.4)**	**(902.3)**	**(898.8)**	**(885.8)**
Forests	CO_2	(598.1)	(574.2)	(354.8)	(500.5)	(639.2)	(695.9)	(700.2)	(701.2)	(703.9)	(703.9)	(703.9)
Harvested Wood	CO_2	(131.8)	(118.4)	(112.9)	(93.4)	(98.2)	(94.8)	(105.3)	(105.4)	(108.6)	(103.0)	(88.0)
Urban Trees[4]	CO_2	(57.1)	(67.3)	(77.5)	n/a	n/a	n/a	n/a	(87.8)	(89.8)	(91.9)	(93.9)
Net Emissions	***All GHGs***	**(414.2)**	**(350.3)**	**(165.1)**	**(154.0)**	**(318.7)**	**(350.5)**	**(359.1)**	**(441.9)**	**(452.6)**	**(443.2)**	**(424.2)**

Note: Parentheses indicate a net sequestration. Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide; CO_2 is carbon dioxide.

[1]Includes emissions from managed manure during storage and transport before soil application.

[2]Agricultural soil C sequestration includes sequestration on land set aside under the Conservation Reserve Program (CRP), in addition to cultivated mineral and organic soils.

[3]Data interpolated for all years except 2001, 2005 and 2008.

[4]Data not reported for years 2001-2004.

1.3 Strategies for Greenhouse Gas Mitigation

Agriculture and forest management can offset GHG emissions by increasing capacity for carbon uptake and storage in biomass, wood products, and soils. This process is referred to as carbon sequestration. The net flux of CO_2 between the land and the atmosphere is a balance between carbon losses from land use conversion and land management practices, and carbon gains from forest growth and sequestration in soils (IPCC 2001). Improved forest regeneration and management practices such as density control, nutrient management, and genetic tree improvement promote tree growth and enhance carbon accumulation in biomass. In addition, wood products harvested from forests can serve as long-term carbon storage pools. The adoption of agroforestry practices like windbreaks and riparian forest buffers, which incorporate trees and shrubs into ongoing farm operations, represents a potentially large GHG sink nationally. While deforestation is a large global source of CO_2, within the United States, net forestland area has experienced a relatively small net loss of roughly 4.2 million hectares (Kimble et al. 2003). Avoidance of large scale deforestation and adoption of the practices mentioned above have resulted in the forestry sector being a net GHG sink in the United States.

Agricultural practices such as conservation tillage and grassland practices such as rotational grazing can also reduce carbon losses and promote carbon sequestration in agricultural soils. These practices offset CO_2 emissions caused by land use activities such as conventional tillage and cultivation of organic soils. However, strategies intended to sequester carbon in soils can also impact the fluxes of two important non-CO_2 GHGs, N_2O and CH_4. Consequently, the net impact of different management strategies on all three biogenic GHGs must be considered when comparing alternatives (Robertson et al. 2000, Del Grosso et al. 2005). Innovative practices to reduce GHG emissions from livestock include modifying energy content of livestock feed, supplementing feed with agents that reduce CH_4 emissions from digestive processes, and managing manure in controlled systems that reduce or eliminate GHG emissions. For example, anaerobic digesters are a promising technology, whereby CH_4 emissions from livestock waste are captured and used as an alternative energy source. Nitrous oxide emissions from soils can be reduced by precision application of nitrogen fertilizers and use of nitrification inhibitors. These and other practices, many of which have additional benefits beyond GHG emission reductions, are discussed further in this report.

1.4 Purpose of this Report

The U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2008 was developed to include emission estimates for years not included in the U.S. Agriculture and Forestry Greenhouse Gas Inventories: 1990-2001 (USDA 2004) and 1990-2005 (USDA 2008) and to revise estimates for previous years based on improved methodologies. This inventory provides a comprehensive assessment of the contribution of U.S. agriculture (i.e., livestock and crop production) and forestry to the national greenhouse gas emissions inventory. The document was prepared to support and expand on information provided in the official Inventory of U.S. GHG Emissions and Sinks (U.S. GHG Inventory), which is prepared annually by the U.S. Environmental Protection Agency to meet U.S. commitments under the United Nations Framework Convention on Climate Change (UNFCCC) (EPA 2010). This report, the U.S. Agriculture and Forestry GHG Inventory (USDA GHG Inventory) supplements the U.S. GHG Inventory, providing an in depth look at agriculture and forestry emissions and sinks of GHG and presenting additional information on GHG emissions from fuel consumption on U.S. farms and ranches. The methodologies and emissions reported here are consistent with the EPA (2010) inventory. There are, however, important differences in reporting that should be noted and understood by the reader. For clarity, two examples of these differences will be noted. First, for the EPA inventory, source and sink categories are defined by the UNFCCC. Because of this, CO_2 emissions from agricultural soils are included in the Land Use, Land Use Change, and Forestry (LULUCF) chapter instead of the Agriculture chapter. In this report, CO_2 emissions from grazed and cropped soils are included in the Livestock and Grazed Land Emissions and Cropland Agriculture chapters, respectively. Second, how energy is distributed among economic sectors is context dependent. This report includes emissions from on-farm energy use, but not the energy emissions associated with the production and transport of farm inputs. The UNFCCC submission (EPA, 2010) combines on-farm energy use with energy associated with farm inputs and reports it in the chapter with energy use from other sectors. Finally, readers should be aware of total versus net emissions. For example, in 2008 total emissions for the United States were 6,957 Tg CO2 eq., but net emissions were 6,016 Tg CO2 eq. after accounting for carbon storage in forests, harvested wood products, and agricultural soils.

The U.S. GHG Inventory provides national-level estimates of emissions of the primary long-lived GHGs (carbon dioxide, methane, nitrous oxide, and fluorinated gases) across a broad range of sectors (energy, industrial processes, solvent use, agriculture, land use change and forestry, and waste). Due to the national-level scale of reporting in the U.S. GHG Inventory, that report does not always provide regional or state GHG emissions data. However, in some cases county, state, and regional emissions data are part of the inventory development process and can be used for more disaggregated analyses.

This report customizes the data from the U.S. GHG Inventory in a manner that is useful to agriculture and forestry producers and related industries, natural resource and agricultural professionals, as well as technical assistance providers, researchers, and policymakers. The information provided in this inventory will be useful in improving our understanding of the magnitude of GHG emissions by county, state, region, and land use, and by crop, pasture, range, livestock and forest management systems. The potential to mitigate emissions from cropped soils is also quantified in this edition of the inventory. The analyses presented in this report are the result of a collaborative process and direct contributions from EPA, USDA (Forest Service, Natural Resources Conservation Service, Agricultural Research Service, Office of Energy Policy and New Uses, and the Climate Change Program Office), and the Natural Resources Ecology Laboratory (NREL) of Colorado State University.

USDA administers a portfolio of conservation programs that have multiple environmental benefits, including reductions in GHG emissions and increases in carbon sequestration. This and future USDA GHG Inventory reports will facilitate tracking of progress in promoting carbon sequestration and reducing GHG emissions through agriculture and forest management. The USDA GHG Inventory describes the role of agriculture and forestry in GHG emissions and sinks, including quantitative estimates of GHG emissions reductions and carbon sequestration through agriculture and forest management. Extensive and indepth emissions estimates are presented for all agricultural and forestry GHG sources and sinks for which internationally recognized methods are available. Where possible, emissions estimates are provided at county, state and regional scales in addition to the national levels provided in the U.S. GHG Inventory. Emissions are categorized by additional information such as land ownership and management practices where possible. This report will help to:

- Quantify current levels of emissions and sinks at county, state, regional, and national scales in agriculture and forestry,
- Identify activities that are driving GHG emissions and sinks and trends in these activities,

- Quantify the uncertainty associated with GHG emission and sink estimates, and
- Quantify the mitigation potential of land management practices intended to reduce GHG emissions.

1.5 Overview of the Report Structure

The report provides detailed trends in agriculture and forestry GHG emissions and sinks, with information by source and sink at county, state and regional levels. The report is structured mainly from a land use perspective, addressing livestock operations, croplands, and forests separately, but it also includes a chapter on energy use. The livestock chapter inventories GHG emissions from livestock and livestock waste stored and managed in confined livestock operations as well as pasture and range operations. The cropland agriculture chapter addresses emissions from cropland soil amendments, rice production, and residue burning, as well as carbon sequestration in agricultural soils. The forest chapter details carbon sequestration in forest biomass and soils, urban trees, and wood products. Fluxes of methane and nitrous oxide in forestry are not addressed since little information is currently available to develop estimates for these sources for forests. Qualitatively, forest soils are net methane sinks in the United States and soil N_2O emissions are small because forests do not receive large N additions (<1% of N fertilizer nationally is applied to forest soils, EPA, 2010). The energy chapter provides information on carbon dioxide emissions from energy consumption on U.S. farms, covering GHG emissions from fuel use in livestock and cropland agriculture. While the U.S. GHG Inventory provides estimates of GHG emissions from energy consumption in the production of fertilizer, this indirect source of agricultural GHG emissions is not covered in this report.

Chapters 2 through 5 present a summary of sources of GHG emissions and sinks in the land use or category of emissions covered by each chapter. A summary of GHG emissions at the national level is provided initially, followed by more detailed descriptions of emissions by each source at national and sub-national scales where available. Methodologies used to estimate GHG emissions and quantify uncertainty are summarized. Changes from the second edition of this inventory are indicated. Text describing the methods and uncertainty for some chapters is summarized from the U.S. GHG Inventory, with permission from the EPA.

1.6 Summary of Changes and Additions for the Third Edition of the Inventory

Compared to previous years, more sophisticated methodologies were used in this report to estimate GHG fluxes form all the major categories. When adjustments are made to existing methodologies (e.g., using new data sources), recalculations are made for the entire time series of estimates to ensure consistency. In addition to updating GHG flux estimates for 1990-2005 (based on current methodologies), estimates for 2006-2008 are also included.

Major changes impacting livestock emissions involved revising animal population estimates or diet assumptions, refining the models used to calculate emissions, using updated activity data, applying anima specific emissions factors, and accounting for sources previously neglected (see Chapter 2 for details). Methane conversion rate, digestible energy values for cattle, and feedlot diets were updated. As a result of these changes, emissions from enteric fermentation increased by approximately 18% on average compared to the previous inventory (USDA 2008). The biggest change for emissions from managed livestock waste is that the inventory now includes indirect N_2O emissions associated with N losses from volatilization of N as ammonia (NH_3), nitrogen oxides (NOx), and leaching and runoff, as recommended by IPCC (2006). These indirect N_2O emissions are added to the direct N_2O emissions to present a more complete picture of N_2O emissions from manure management. As a result of this change in methodology, N_2O emission estimates from manure management systems have increased by approximately 60 percent compared to the previous inventory. In this edition, N additions to soils from grazing animals are consistent with N excretion data. Nitrate leaching was assumed to be an insignificant source of indirect N_2O in grassland systems where the amount of precipitation plus irrigation did not exceed the potential evapotranspiration, as recommended by IPCC (2006). These changes resulted in an approximately 40 percent decrease in grazed soil N_2O emissions. The biggest change that impacted estimates of carbon dioxide fluxes for grazed lands involved using annual survey data from the USDA National Resources Inventory (NRI). Availability of new data extended the time series of activity data beyond 1997 to 2003. In previous inventories, activity data were only available through 1997 at 5-year intervals, and subsequent years were treated as the same land use practice occurring in 1997. NRI area data were reconciled with the forest area estimates in the Forest Inventory and Analysis (FIA) dataset, and were incorporated into the estimation of soil C stock changes. These changes resulted in an average annual increase in C stocks of soils used for livestock grazing of approximately 40 Tg CO_2 eq. for the time series, compared to the previous Inventory.

Although there were no major changes in methodologies for cropland emissions (Chapter 3) compared to the previous edition (USDA 2008), a series of improvements were implemented. Instead of assuming that nitrate leaching can occur anywhere, a criterion was used to designate lands where nitrate is susceptible to be leached into waterways, as suggested by IPCC (2006). Other changes include: using state-level N data for on-farm use of fertilizers to estimate synthetic N fertilizer application on non-major crops, including uncertainty in model outputs of N volatilization and N leaching/runoff in the calculation of uncertainty for indirect emissions, using a default uncertainty of ±50 percent for Tier 1 uncertainties that were not addressed in the previous inventory (e.g., crop yields and organic fertilizer amendments), improved estimates of manure N available for land application, revising the model parameterization for sorghum, and correcting uncertainty calculations. The main results of these changes are lower N_2O emissions and wider confidence intervals.

Estimates of forest carbon stock changes (Chapter 4) reflect a substantial number of incremental changes in methods and data compared to the previous inventory. The accumulation of newer inventory data for most states, including stocks for coastal (southern and eastern) Alaska and western Texas, affect carbon stock totals and changes compared to previous inventories. Updated land area analysis resulted in reduction of grassland area in the United States, because woodlands previously designated as grassland are now considered forest land, thus increasing the estimation of soil C stock changes in these areas. However, redefining forestland also led to the removal of low cover, lower productivity woodlands areas from the surveys (Smith et al. 2009), which were included in the previous USDA (2008) inventory. On average, these changes increased carbon stock estimates by approximately 8 percent.

2.1 Summary of U.S. Greenhouse Gas Emissions from Livestock

A total of 234 Tg CO_2 eq. of greenhouse gases (GHGs) were emitted from livestock, managed livestock waste, and grazed land in 2008 (Table 2-1, Figure 2-1). This represents about 55% of total emissions from the agricultural sector (EPA 2010). Compared to the baseline year (1990), emissions from this source were about 26% higher in 2008. The 95% confidence interval for 2008 was estimated to lie between 204 and 332 Tg CO_2 eq. (Table 2-1).

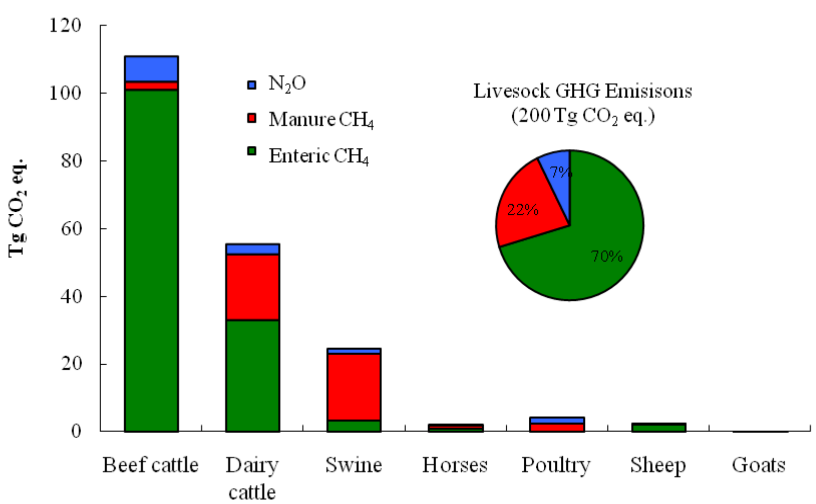
Table 2-1 Greenhouse Gas Emission Estimates and Uncertainty Intervals in 2008

	Estimate	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Source		*Tg CO_2 eq.*		*percent*	
CH_4 enteric fermentation	141	125	166	-11	+18
CH_4 managed waste + grazed land	48	39	57	-18	+20
N_2O managed waste	14	12	18	-16	+24
N_2O grazed land	62	39	156	-37	+153
CO_2 grazed land remaining grazed land	(5)	(7)	(3)	-53	+42
CO_2 land converted to grazed land	(27)	(29)	(24)	-8	+9
Total	**234**	**204**	**332**	**-13**	**+42**

Note: Parentheses indicate a net sequestration. Tg CO_2 eq. is teragrams carbon dioxide equivalent. CH_4 is methane. N_2O is nitrous oxide. CO_2 is carbon dioxide.

Enteric fermentation was responsible for over half (141 Tg CO_2 eq.) of all emissions associated with livestock production, while grazed lands (62 Tg CO_2 eq.) and managed waste (48 Tg CO_2 eq.) accounted for approximately 26% and 20% of the total emissions. All of the emissions from enteric fermentation and about 77% of emissions from managed livestock waste were in the form of methane (CH_4). Of the emissions from grazed lands, 96% were in the form of nitrous oxide (N_2O) from soils (Table 2-2). Soils in grazed lands do not often experience the anaerobic conditions required for CH_4 production to exceed CH_4 uptake. However, a small portion of manure from grazing animals is converted to CH_4. Grazed lands served as a sink for CO_2 emissions, sequestering 31.4 Tg in 2008 (Table 2-2). The largest total emissions associated with livestock production were from

Figure 2-1
Greenhouse Gas Emissions from Livestock in 2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Texas and California (Map 2-1). Emissions were high in Texas primarily because of the large numbers of beef cattle, while dairy cattle emissions are responsible for most emissions in California. Emissions were also high in Iowa, Nebraska, Kansas, Oklahoma, and Missouri. Beef cattle were responsible for the largest fraction (55%) of GHG emissions from livestock in 2008, with the majority of emissions in the form of CH_4 from enteric fermentation and N_2O from grazed land soils (Figure 2-1, Table 2-2). Dairy cattle were the second largest livestock source of GHG emissions (28%), primarily CH_4 from enteric fermentation and managed waste. The third largest GHG source from livestock was swine (12%), nearly all of which was CH_4 from waste. Horses, goats, and sheep caused relatively small GHG emissions when compared to other animal groups, because populations of these types are relatively small.

Table 2-2 Greenhouse Gas Emissions by Livestock Category and Source in 2008

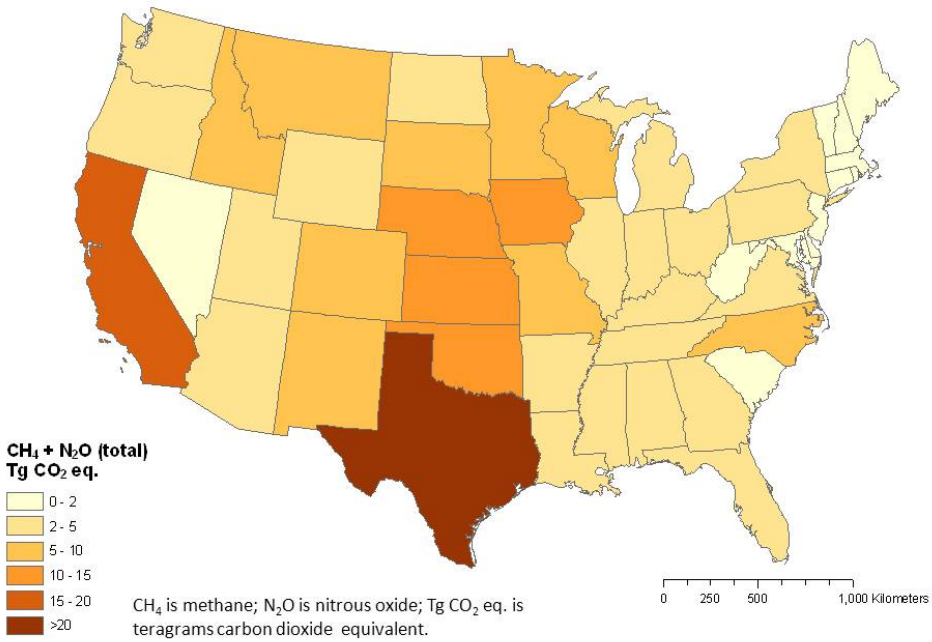
	Enteric Fermentation	Managed Livestock Waste		Grazed Land			
	CH_4	CH_4	N_2O	N_2O[1]	CH_4	CO_2	***Total***
Animal Type			*Tg CO_2 eq.*				
Beef cattle	100.77	2.47	7.44	51.90	1.97	(26.40)	**138.2**
Dairy cattle	33.09	19.43	5.48	1.68	0.05	(0.85)	**58.9**
Swine	3.59	19.58	1.65	0.20	0.01	(0.10)	**24.9**
Horses	1.00	0.82	0.41	6.92	0.76	(3.52)	**6.4**
Poultry	0.00	2.63	1.77	0.12	0.01	(0.06)	**4.5**
Sheep	2.12	0.08	0.34	0.51	0.04	(0.26)	**2.8**
Goats	0.27	0.02	0.02	0.39	0.02	(0.20)	**0.5**
Total	**140.8**	**45.0**	**17.1**	**61.7**	**2.85**	**(31.4)**	**236.2**

Note: Parentheses indicate a net sequestration. Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide; CO_2 is carbon dioxide.
[1]Includes direct and indirect emissions.

Livestock contribute GHGs to the atmosphere both directly and indirectly. Livestock emit CH_4 directly as a byproduct of digestion through a process called enteric fermentation. In addition, livestock manure and urine ("waste") cause CH_4 and N_2O emissions to the atmosphere through increased decomposition and nitrification/denitrification. Managed waste that is collected and stored emits CH_4 and N_2O. Grazing animals influence soil processes (nitrification/denitrification) that result in N_2O emissions from the nitrogen (N) in their waste, which increases N_2O emissions. Forage legumes on grazed lands also contribute to N_2O emissions because legumes fix nitrogen from the atmosphere which can become mineralized in the soil and contribute to nitrification and denitrification. Grazed lands can also act as a source or sink for atmospheric carbon dioxide (CO_2), depending on whether carbon inputs to the soil from plant residues and manure exceed carbon losses from decomposition of soil organic matter. Soils that have been historically cropped using conventional tillage are often depleted of carbon because tillage disturbs soil aggregates and warms soil, both of which increase decomposition rates. Carbon depleted soils can act as CO_2 sinks upon conversion to grazing because grazed soils are typically not plowed. Factors such as grazing intensity and weather patterns also influence net CO_2 fluxes, so a particular parcel of grazed land may be a net source or sink of carbon during any given year.

This chapter provides national and state-level data on CH_4 emissions from enteric fermentation, CH_4 and N_2O emissions from managed livestock waste, and CO_2, N_2O and CH_4 fluxes for grazed lands. Emissions associated with waste applied to grazed land are included in this chapter, while nitrous oxide emissions from managed livestock waste applied to cropped soils are included in the Cropland Agriculture chapter (Chapter 3). State-level livestock population data also are presented in this chapter because GHG emissions from livestock are related to livestock population sizes.

Map 2-1
GHG emissions from livestock in 2008.



CH_4 is methane; N_2O is nitrous oxide; Tg CO_2 eq. is teragrams carbon dioxide equivalent.

2.2 Sources of Greenhouse Gas Emissions from Livestock

The mechanisms and important factors in generating GHG fluxes from livestock, waste management, and grazed lands are detailed below.

2.2.1 Enteric Fermentation

Enteric fermentation is a normal digestive process where anaerobic microbial populations in the digestive tract ferment food and produce CH_4 gas as a byproduct. Methane is then emitted from the animal to the atmosphere through exhaling or eructation. Ruminant livestock, including cattle, sheep, and goats, have greater rates of enteric fermentation because of their unique digestive system, which includes a large rumen or fore-stomach where enteric fermentation takes place. Non-ruminant livestock such as swine, horses, and mules produce less CH_4 from enteric fermentation because it takes place in the large intestine, which has a smaller capacity to produce CH_4 than the rumen. The energy content and quantity of animal feed also affect the amount of CH_4 produced in enteric fermentation, with lower quality and higher quantities of feed causing greater emissions.

2.2.2 Managed Livestock Waste

Livestock waste can be "managed" in storage and treatment systems, or spread on fields in lieu of long-term storage. Alternatively, livestock waste is termed "unmanaged" when it is deposited directly on grazed lands and not transported. Many livestock producers in the U.S. manage livestock waste in systems such as solid storage, dry lots, liquid-slurry storage, deep pit storage, and anaerobic lagoons. Table 2-3 provides descriptions of managed and unmanaged pathways for livestock waste, indicating the relative impacts of different pathways on GHG emissions. Sometimes livestock waste that is stored and treated is subsequently applied as a nutrient amendment to agricultural soils. GHG emissions from the application of treated waste to cropped soils as a nutrient amendment are discussed in the next chapter along with GHG emissions from other nutrient amendments for crop production.

Table 2-3 Descriptions of Livestock Waste Deposition and Storage Pathways

Manure Management System	Description	Relative Emissions CH_4	N_2O
Pasture / Range / Paddock	Manure and urine from pasture and range grazing animals is deposited directly onto the soil.	low	high
Daily Spread	Manure and urine are collected and spread on fields, there is little or no storage of the manure/urine before it is applied to soils.	low	zero[1]
Solid Storage	Manure and urine (with or without litter) are collected by some means and placed under long-term bulk storage.	low	high
Dry Lot	Manure and urine are deposited directly onto unpaved feedlots where the manure is allowed to dry and it is periodically removed (after removal it is sometime spread onto fields).	low	high
Liquid / Slurry	Manure and urine are collected and transported in a liquid state to tanks for storage. The liquid/slurry mixture may be stored for a long-time and water may be added to facilitate handling.	moderate to high	low
Anaerobic Lagoon	Manure and urine are collected using a flush system and transported to lagoons for storage. Manure/urine resides in lagoons for 30-200 days.	variable	low
Pit Storage	Combined storage of manure and urine in pits below livestock confinements.	moderate to high	low
Poultry with Litter	Enclosed poultry houses use bedding derived from wood shavings, chopped straw, or other products depending on availability. The bedding absorbs moisture and dilutes manure. Litter is cleaned out once a year. This system is used for breeder flocks and meat chickens (broilers) and other fowl.	low	high
Poultry without Litter	In high-rise cages or scrape-out/belt systems, manure is excreted onto the floor below with no bedding to absorb moisture. The ventilation system dries the manure as it is stored. This high-rise system is a form of passive windrow composting.	low	low

Adapted from IPCC (2000) Chapter 4. CH_4 is methane; N_2O is nitrous oxide.

[1]N_2O emissions are assumed to be zero during the transport/storage phase but not after the waste has been applied to soils.

The magnitude of CH_4 and N_2O emissions from managed livestock waste depends in large part on environmental conditions. Methane is emitted under anaerobic conditions, when oxygen is not available to the bacteria which decompose waste. Storage in ponds, tanks, or pits such as those that are coupled with liquid/slurry flushing systems often promote anaerobic conditions (i.e., where oxygen is not available and CH_4 is produced), whereas solid waste stored in stacks or shallow dry pits tends to provide aerobic conditions (i.e., where oxygen is available and little or no CH_4 is produced). High temperatures generally accelerate the rate of decomposition of organic compounds in waste, increasing CH_4 emissions under anaerobic conditions. In addition, longer residency time in a storage system can increase CH_4 production, and moisture additions, particularly in solid storage systems that normally experience aerobic conditions, can amplify CH_4 emissions.

While environmental conditions are important factors affecting CH_4 emissions from the management of livestock waste, diet and feed characteristics are also influential. Livestock feed refers to the mixture of grains, hay and byproducts from processed foods that is fed to animals at feedlots and supplemental feed for grazing animals, while diet includes the mixture of plants that animals graze. Livestock feed, diet, and growth rates affect both the amount and quality of manure. Not only do greater amounts of manure lead to higher CH_4 production, but higher energy feed also produces manure with more volatile solids, increasing the substrate from which CH_4 is produced. However, this impact is somewhat offset because some higher energy feeds are more digestible than lower quality forages, and thus less waste is excreted.

The production of N_2O from managed livestock waste depends on the composition of the waste, the type of bacteria involved, and the conditions following excretion. For N_2O emissions to occur, the waste must first be handled aerobically where ammonia or organic nitrogen is converted to nitrates and nitrites (nitrification), and if conditions become sufficiently anaerobic, nitrates and nitrites can be denitrified, i.e., reduced to N oxides and nitrogen gas (N_2) (Groffman et al. 2000). Nitrous oxide is produced as an intermediate product of both nitrification and denitrification and can be directly emitted from soil as a result of both of these processes. These emissions are most likely to occur in dry waste handling systems that have aerobic conditions, but that also contain pockets of anaerobic conditions due to high water contents and high oxygen gas (O_2) demand from decomposition. For example, waste in dry lots is deposited on soil, is oxidized to nitrite and nitrate, and encounters anaerobic conditions following precipitation events that increase water content, enhance decomposition, and deplete the supply of O_2.

Managed livestock waste can also contribute to indirect N_2O emissions. Indirect emissions result from nitrogen that was emitted or leached from the manure management system in a form other than N_2O and was then converted to N_2O offsite. These sources of indirect N_2O emission from animal waste are from ammonia (NH_3) volatilization, nitric oxide (NO) emissions from nitrification and denitrification, and nitrate (NO_3) leached or run off into ground or surface waters. The gaseous losses of NH_3 and NO to the atmosphere can then be deposited to the soil and converted to N_2O by nitrification. The nitrate leached or run off into waterways can be converted to N_2O by aquatic denitrification.

2.2.3 Grazed Lands

Nitrous oxide from soils is the primary GHG associated with grazed lands. Grazed lands contribute to N_2O emissions by adding nitrogen to soils from animal wastes and from forage legumes. Legumes fix atmospheric N_2 into forms that can be used by plants and by soil microbes. Nitrogen from manure and legumes is cycled into the soil and can provide substrates for nitrification and denitrification. Nitrous oxide is a by-product of this cycle; thus more nitrogen added to soils yields more N_2O released to the atmosphere. A portion of the nitrogen cycled within the plant-animal-soil system volatilizes to the atmosphere in various gaseous forms and is eventually re-deposited onto the soils where it can contribute to indirect N_2O emissions. Some nitrogen in the form of nitrate can leach into groundwater and surface runoff, undergo denitrification, and contribute to indirect N_2O emissions. In addition to nitrogen additions, weather, soil type, grazing intensity and other factors influence emissions from grazed lands.

Manure deposited on grazed lands also produces CH_4 emissions. Methane emissions from this source are relatively small, less than 3% of total grazed land GHG emissions, because of the predominately aerobic conditions that exist on most pastures and ranges.

Grazed lands can be emission sources or net sinks for CO_2. Typically, cropland that has recently been converted to grazed land stores CO_2 from the atmosphere in the form of soil organic carbon. But after sufficient time, soil organic carbon reaches a steady state, given consistent weather patterns. Long-term soil carbon levels are sensitive to climate change and soils that were previously sinks can revert to being sources of CO_2.

2.3 U.S. Livestock Populations

Greenhouse gas emissions from livestock are related to population size. Livestock population data are collected annually by USDA's National Agricultural Statistics Service (NASS). Those data are an input into the GHG estimates from livestock in the U.S. GHG Inventory.

Beef and dairy cattle, swine, sheep, goats, poultry, and horses are raised throughout the United States. Detailed livestock population numbers for each state in 2008 are provided in Appendix Table A-1. Appendix Table A-2 shows total national livestock population sizes from 1990 to 2008 by livestock categories. Trends for beef cattle, dairy cattle, and swine are described in more detail below because of their relatively high population numbers and consequently high contributions to GHG emissions.

Texas raised by far the most beef cattle, at over 14 million head in 2008 (Appendix Table A-1). Kansas, Nebraska, Oklahoma, Iowa, and Missouri each raised from 4 to 6 million head of beef cattle, while several other states raised approximately (~)2 million head. Fewer dairy cattle than beef cattle are raised currently in the United States. Dairy cattle populations were highest in California (~2.6 million) and Wisconsin (~1.9 million) (Appendix Table A-1). New York, Idaho, Pennsylvania, and Minnesota had the next largest populations of dairy cattle, ranging from 730,000 to 970,000 head in each state. Most states had fewer than 500,000 head of dairy cattle.

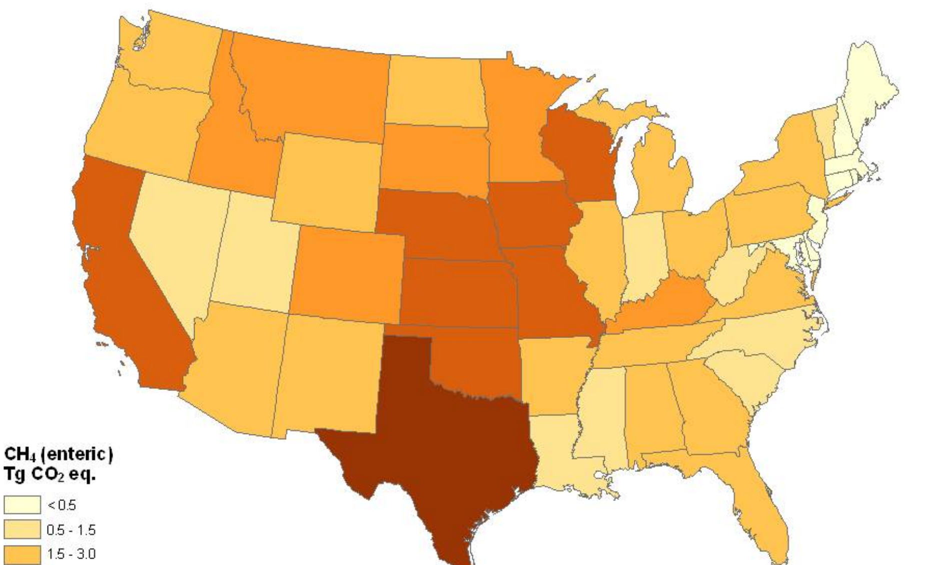
Iowa was the largest swine producer with 19.5 million head in 2008 (Appendix Table A-1). North Carolina housed the second largest swine population at 10 million head. Illinois, Indiana, Minnesota, Missouri, Nebraska, and Oklahoma also have sizeable swine populations.

Arkansas and Georgia had the largest poultry populations in 2008, with roughly 240 million and 280 million head of poultry in each state, respectively (Appendix Table A-1). Alabama, North Carolina, Mississippi, and Texas also had large populations of poultry, between 141 and 209 million head each. Iowa, Indiana, Kentucky, Maryland, and Virginia had poultry populations between 51 and 84 million head.

2.4 Enteric Fermentation

Just over half (53%) of emissions associated with livestock production were from CH_4 produced by enteric fermentation. Cattle were responsible for the vast majority of enteric CH_4 emissions (95%) in 2008 (Table 2-2). Texas (17.9 Tg CO_2 eq.) and California (9.1 Tg CO_2 eq.) had the largest CH_4 emissions from enteric fermentation for beef cattle and dairy cows in 2008 (Map 2-2, Appendix Table A-4). These emissions were largely tied to the sizable populations of cattle in both states. However, enteric fermentation emissions in Texas were mostly from beef cattle, whereas in California they were mostly from dairy cattle (Appendix Table A-4). State-level data for non-cattle livestock (i.e., swine, sheep, goats, and horses) or bulls was not generated due to the relatively low contributions of these animals to total enteric emissions. Central, Northern Plains, and some Great Lakes states also had relatively high CH_4 emissions from enteric fermentation, ranging between 3 and 8.5 Tg CO_2 eq. per state in 2008 (Appendix Table A-4). Emissions tended to be lower from some states in the Northeast, Southeast, and the desert Southwest, mainly because cattle populations are low in these states.

Map 2-2
Methane emissions from enteric fermentation in 2008.



CH_4 is methane; Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Annual emissions of CH_4 from enteric fermentation fluctuated by approximately 10 Tg CO_2 eq. between 1990 and 2008 (Table 2-4). Emissions peaked in 1995, then decreased by about 10 Tg CO_2 eq. by 2005 and were back up near 1995 emissions by 2008. Overall, by 2008, CH_4 emissions from enteric fermentation increased by about 6% compared to 1990 levels.

2.4.1 Methods for Estimating Methane Emissions from Enteric Fermentation

The official U.S. GHG Inventory estimates for enteric fermentation are calculated according to the methodological framework provided by the Intergovernmental Panel on Climate Change (IPCC) for preparing national GHG inventories. The IPCC guidance is

Table 2-4 U.S. Methane Emissions from Enteric Fermentation in 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Animal Type						*Tg CO_2 eq.*					
Beef cattle	94.5	107.7	100.6	99.9	100.0	100.0	98.3	99.3	100.9	101.6	100.8
Dairy cattle	32.0	30.5	30.9	30.7	30.8	28.7	30.1	30.6	31.3	32.7	33.1
Horses	1.9	1.9	2.0	2.1	2.3	2.6	3.0	3.5	3.6	3.6	3.6
Sheep	1.9	1.5	1.2	1.2	1.1	1.1	1.0	1.0	1.0	1.0	1.0
Swine	1.7	1.9	1.9	1.9	1.9	1.9	1.9	1.9	1.9	2.1	2.1
Goats	0.3	0.2	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3
Total	**132.4**	**143.7**	**136.8**	**136.0**	**136.3**	**134.5**	**134.6**	**136.7**	**139.0**	**141.2**	**140.8**

Tg CO_2 eq. is teragrams carbon dioxide equivalent.

organized into a hierarchical, tiered analytical structure, in which higher tiers correspond to more complex and detailed methodologies. The methods detailed below correspond to both Tier 1 and Tier 2 approaches. With the permission of EPA, Annex 3.9 from the official U.S. GHG Inventory is summarized below. Methane emissions from enteric fermentation were estimated for five livestock categories: cattle, horses, sheep, swine, and goats. Emissions from cattle represent the majority of U.S. emissions; consequently, the more detailed IPCC Tier 2 methodology was used to estimate emissions from cattle, and the IPCC Tier 1 methodology was used to estimate emissions from the other types of livestock.

2.4.1.1 Estimating Methane Emissions from Cattle

This section describes the process used to estimate enteric fermentation emissions of CH_4 from cattle on a regional basis. A Cattle Enteric Fermentation Model (CEFM) based on recommendations provided in IPCC (2006, 1997) was developed that uses information on population, energy requirements, digestible energy, and the fraction of energy converted to methane to estimate CH_4 emissions. The emission estimation methodology consists of the following three steps: (1) characterize the cattle population to account for cattle population categories with different emissions profiles; (2) characterize cattle diets to generate information needed to estimate emissions factors; and (3) estimate emissions using these data and the IPCC Tier 2 equations.

Step 1: Characterize U.S. Cattle Population

Each stage in the cattle lifecycle was modeled to simulate the cattle population from birth to slaughter. This level of detail accounts for the variability in CH_4 emissions associated with each life stage. Given that the time in which cattle can be in a stage can be less than 1 year (e.g., beef calves are weaned at 7 months), the stages are modeled on a per-month basis. The type of cattle use also impacts CH_4 emissions (e.g., beef versus dairy). Consequently, cattle life stages were modeled for several categories of dairy and beef cattle. These categories are listed in Appendix Table A-5.

The key variables tracked for each of these cattle population categories[1] includes calving rates, pregnancy and lactation (Appendix Table A-6), average weights and weight gains (Appendix Table A-7), feedlot placements (Appendix Table A-8), death rates, number of animals per category each month, and animal characteristics (i.e., age, gender, etc.) data.

Cattle population data were taken from USDA National Agricultural Statistics Survice (NASS) (Appendix Table A-2). The USDA NASS publishes monthly, annual, and multi-year livestock population and production estimates. Multi-year reports include revisions to earlier published data. Cattle and calf populations, feedlot placement statistics (e.g., number of animals placed in feedlots by weight class), slaughter numbers, and lactation data were obtained from the USDA NASS (Cattle: USDA NASS 2004, 1999). Beef calf birth percentages were obtained from the USDA Animal and Plant Health Inspection Service (APHIS) National Animal Health Monitoring System (USDA APHIS NAHMS 2008, 1997).

Step 2: Characterize U.S. Cattle Diets
To support development of digestible energy (DE), the percent of gross energy intake digestible to the animal and CH_4 conversion rate (Y_m) (i.e., the fraction of gross energy converted to CH_4 values for each of the cattle population categories) data were collected on diets considered representative of different regions. For both grazing animals and animals being fed mixed rations, representative regional diets were estimated using information collected from state livestock specialists and from USDA APHIS NAHMS (2008). The data for each of the diets (e.g., proportions of different feed constituents, such as hay or grains) were used to determine chemical composition for use in estimating DE and Y_m for each animal type. Region- and cattle-type-specific estimates for DE and Y_m were developed for the U.S. (Appendix Table A-9). Regions are defined in (Appendix Table A-10). Additional detail on the regional diet characterization is provided in EPA (2010).

Step 3: Estimate Methane Emissions from Cattle

Emissions were estimated in three steps: (a) determine gross energy intake using the IPCC (2006) equations, (b) determine an emissions factor using the DE values and other factors, and (c) sum the daily emissions for each animal type. The necessary data values include:

- Body weight (kg)
- Weight gain (kg/day)
- Net energy for activity (Mj/day)
- Standard reference weight (dairy = 1,324 lbs; beef = 1,195 lbs)
- Milk production (kg/day)

[1] Except bulls. Only end-of-year census population statistics and a national emission factor are used to estimate CH_4 emissions from the bull population.

- Milk fat (% of fat in milk = 4)
- Pregnancy (% of population that is pregnant)
- DE (% of gross energy intake digestible)
- Y_m (the fraction of gross energy converted to CH_4)

This process was repeated for each month, and the totals for each subcategory were summed to achieve an emissions estimate for the entire year. The estimates for each of the 10 subcategories of cattle are listed in Appendix Table A-11. The CH_4 emissions for each subcategory were then summed to estimate total emissions from beef cattle and dairy cattle for the entire year. The cattle emissions calculation model estimates emissions on a regional scale. Individual state-level estimates were developed from these regional estimates using the proportion of each cattle population subcategory in the state relative to the population in the region.

2.4.1.2 Emission Estimates From Other Livestock

All livestock population data, except for horses, were taken from USDA NASS (1994) reports (Hogs and pigs; Sheep and goats). Appendix Table A-2 shows the population data for all livestock that were used for estimating all livestock-related emissions. For each animal category, the USDA publishes monthly, annual, and multi-year livestock population and production estimates. Multi-year reports include revisions to earlier published data. Recent reports were obtained from the USDA Economics and Statistics System, while historical data were downloaded from USDA NASS. The Food and Agriculture Organization (FAO) of the United Nations publishes horse population data. These data were accessed from the FAOSTAT database (FAO 2009). National-level emission calculations for other livestock were developed from national population totals. State-level emissions for each livestock type were developed from these national totals based on the proportion of livestock population in each state relative to the national total population for the particular livestock category and by assuming that emissions are proportional to populations. Appendix Table A-12 shows the emission factors used for these other livestock.

2.4.2 Uncertainty in Estimating Methane Emissions from Enteric Fermentation

The following discussion of uncertainty in the enteric fermentation estimates is from the U.S. GHG Inventory (EPA 2010) and reproduced here with permission from EPA.

Uncertainty is estimated using the Monte Carlo Stochastic Simulation technique. Emission factors and animal population data are the primary sources of uncertainty in estimating CH_4 emissions from enteric fermentation. One hundred eighty-five input variables were identified as key input variables for uncertainty analysis (e.g., estimates of births by month, weight gain of animals by age class, and placement of animals into feedlots based on placement statistics and slaughter weight data). The uncertainty associated with these input variables is ±10% or lower. However, the uncertainty for many of the emission factors is over ± 20%. The overall 95% confidence interval around the estimate of 141 Tg CO_2 eq. ranges from 125 to 166 Tg CO_2 eq. (Table 2-1).

2.4.3 Changes Compared to the 2nd Edition of the USDA GHG Report

There were several important modifications made to the emissions estimates for this edition of the USDA GHG report relative to the previous Inventory (USDA 2008b). Most of the changes involved revising animal population estimates or diet assumptions, or refining the model used to calculate emissions. Heifer and steer stocker populations previously left out of the emissions calculations are now included, and beef, dairy, swine, and horse populations were revised. The FAO horse population estimates increased dramatically between the current and previous Inventory. Enteric fermentation data for bull populations are no longer averaged between January and July because of the high degree of uncertainty related to July estimates, so populations are based solely on January estimates. An adjustment was made to the CEFM to allow feedlot placements for the 700–800 lbs category to use excess animals from the over 800 lbs category if insufficient animals are available to place in a given month at 700–800 lbs. Calf weight at 7 months was adjusted to be equal for all months, as current research indicated that evidence was not sufficient to suggest that calf weight at weaning differs by birth month. Mature weight for beef cows was revised based on annual data collected from 1989 through 2007, as was replacement weight at 15 and 24 months. Mature weight for dairy cows was adjusted to 1,550 for all years, and replacement weight at 15 and 24 months was adjusted accordingly. Monthly weight gain for stockers and coefficients used for calculating the net energy required for maintenance used for lactating cattle were increased.

Methane conversion rate (Ym) and digestible energy (DE) values for cattle were updated based on model evaluations (Kebreab et al. 2008) and literature values. Feedlot diets were updated based on current survey data from Galyean and Gleghorn (2001) and Vasconcelos and Galyean (2007). Further modifications were made to feedlot placement methodology when discrepancies existed between simulated USDA placement data for weight class and number of calculated animals available by weight. The models would account for these differences by pulling available stockers from the higher weight category. If a minimum number of available stockers still could not be reached, animals were pulled from the next lower weight category.

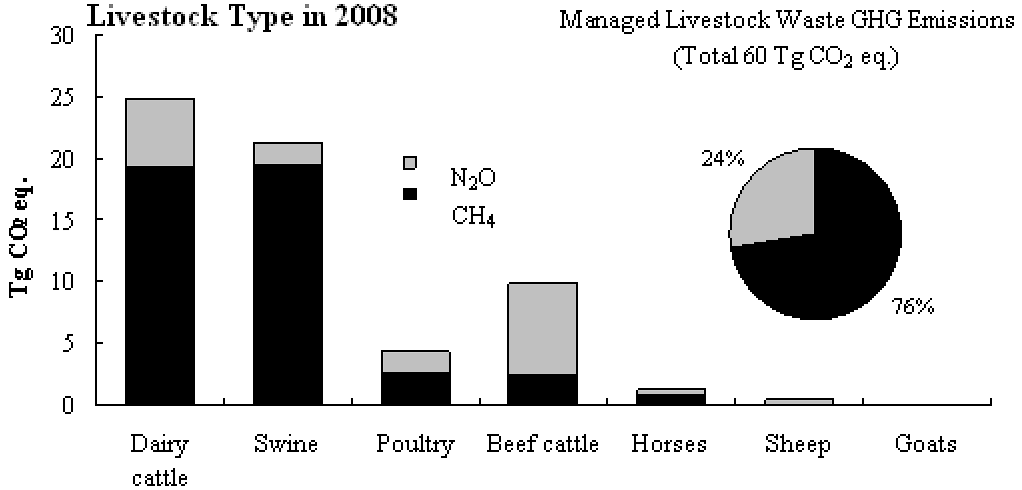
As a result of the changes outlined above, emissions from enteric fermentation increased by approximately 18% on average compared to the previous Inventory (USDA 2008).

2.5 Managed Livestock Waste

Greenhouse gas emissions from managed livestock waste are composed of CH_4 and N_2O from livestock waste storage and treatment and CH_4 emissions from the daily spread of livestock waste. Emissions from these sources are discussed below, with estimates disaggregated spatially and by livestock category where possible. Methane was the predominant GHG emitted from managed livestock waste in 2008, accounting for 72% of 62 Tg CO_2 eq. total emissions from this source (Table 2-5). The remaining 28% of GHG emissions from managed livestock waste was N_2O. Dairy cattle and swine were each responsible for 37% and 36% of total managed waste emissions respectively (Figure 2-2). Poultry (7%) and beef cattle (17%) were also important sources in 2008. For beef cattle, N_2O was the predominate form (75%) of waste emissions. Over time, emissions from managed waste increased by ~40% from 1990 to 2008 (Figure 2-3). Most of the increase was from higher CH_4 emissions due to the trend of storing more waste in liquid systems and anaerobic lagoons, which facilitate CH_4 production.

Figure 2-2

Greenhouse Gas Emissions from Managed Livestock Waste by Livestock Type in 2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent

Table 2-5 Greenhouse Gas Emissions from Managed Livestock Waste in 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
GHG Type	*Tg CO_2 eq.*[3]										
Nitrous Oxide[1]	14.4	15.5	16.7	16.5	16.8	16.3	16.4	16.6	17.3	17.3	17.1
Methane[2]	29.3	33.9	38.6	40.1	41.2	38.4	40.2	42.2	42.3	45.9	45.0
Total	**43.7**	**49.3**	**55.2**	**56.6**	**57.9**	**54.7**	**56.5**	**58.9**	**59.6**	**63.2**	**62.1**

[1] Does not include emissions from managed manure applied to cropped soils.

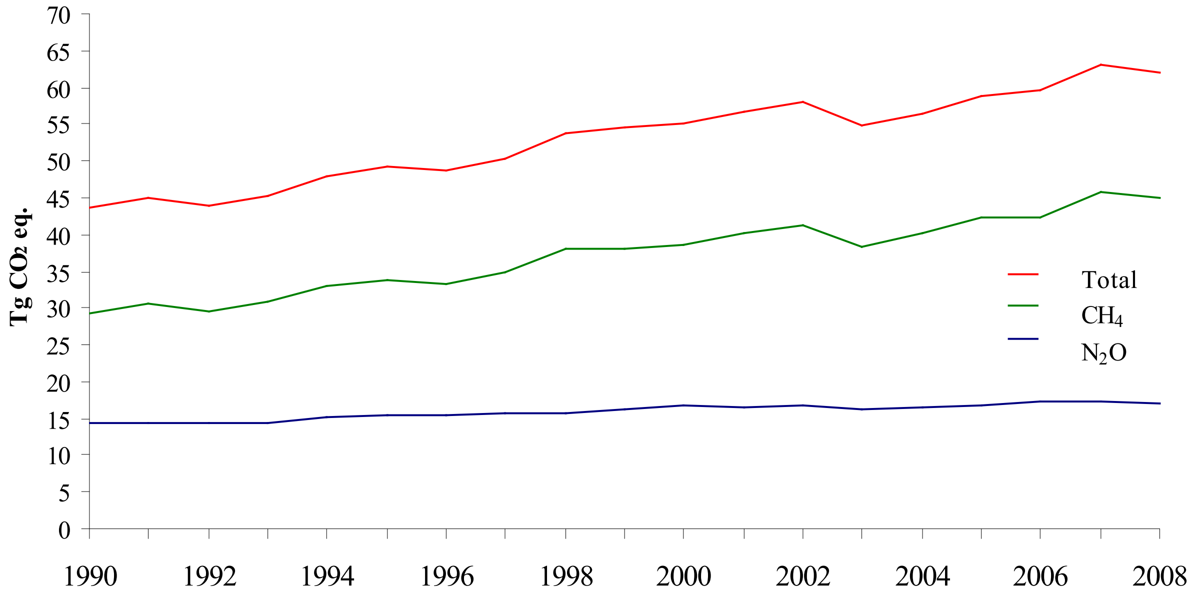
[2] Includes CH_4 from managed sources and from grazed grasslands. Manure deposited on grasslands produces little CH_4 due to predominantly aerobic conditions

[3] Teragrams carbon dioxide equivalent.

While beef cattle are responsible for the largest overall emissions from all livestock, (Table 2-2, Figure 2-1), emissions from beef cattle managed waste are relatively small (Figure 2-2) because most waste generated by beef cattle is unmanaged. Emissions from beef cattle managed manure changed little between 1990 and 2008. Managed manure emissions from horses, sheep, and goats are small due to the relatively small population of these animals (Appendix Table A-2), and, as for beef cattle, most of the manure is unmanaged or managed in dry systems (EPA 2010). State-level GHG emissions from managed livestock waste varied across states in 2008, with a small number of states responsible for the larger contributions to national GHG emissions. California and Iowa had the largest GHG emissions from managed livestock waste (4.8 and 4.6 Tg CO_2 eq., respectively; Map 2-3). In North Carolina, this was primarily from swine. In Texas, however, most emissions were from both beef and dairy cattle waste, with a smaller portion from swine (Appendix Table A-14, A-15).

Figure 2-3

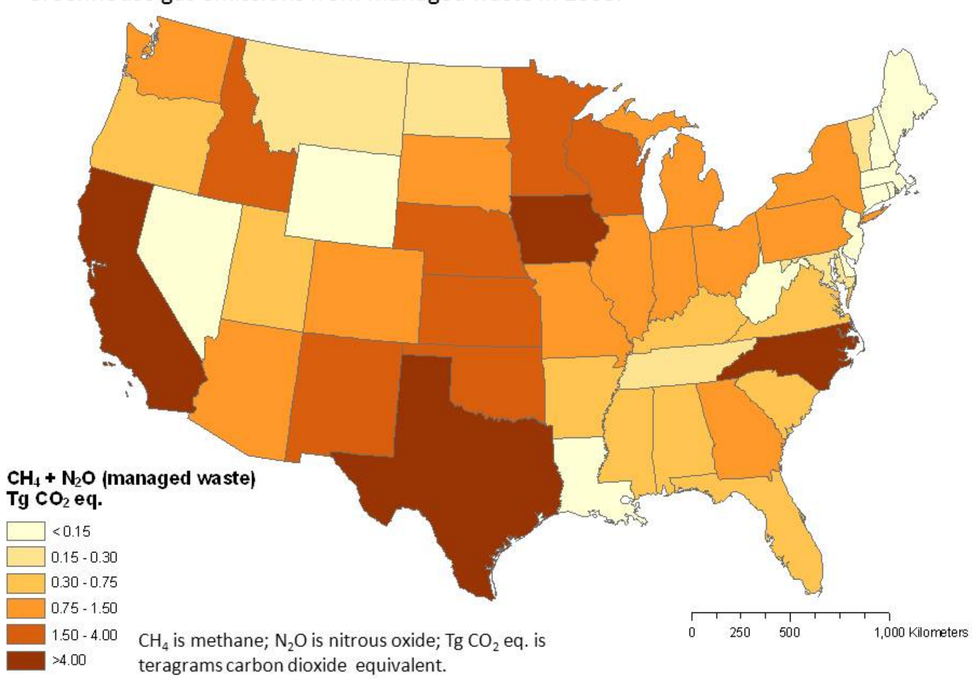
Greenhouse Gas Emissions from Managed Livestock Waste, 1990-2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide.

Map 2-3

Greenhouse gas emissions from managed waste in 2008.



CH_4 is methane; N_2O is nitrous oxide; Tg CO_2 eq. is teragrams carbon dioxide equivalent.

2.5.1 Methods for Estimating Methane and Nitrous Oxide Emissions from Managed Livestock Waste

This section summarizes how CH_4 and N_2O emissions from livestock waste were calculated in the U.S. GHG Inventory (EPA 2010) as well as for this inventory report. Animal population data are used to estimate CH_4 production potential and nitrogen in waste, and these are multiplied by a methane conversion factor (MCF) and direct and indirect N_2O emission factors. MCFs are used to determine the amount of CH_4 emissions that are potentially produced by each unit of livestock waste. MCFs vary by livestock type, manure storage system, and the waste storage temperature. The IPCC (2006) default direct N_2O emission factor was used while indirect N_2O emission factors varied by region and waste management system. The EPA provides the USDA with state and national estimates of GHG emissions from managed livestock waste. The estimates of GHG emissions from managed livestock waste were prepared following a methodology developed by EPA and consistent with international guidance, and are described in detail in Annex 3.10 of the U.S. GHG Inventory (EPA 2010).

Data required to calculate CH4 emissions from livestock waste:

- Animal population data (by animal type and state);
- Typical Animal Mass (TAM) data (by animal type);
- Portion of manure managed in each Waste Management System (WMS), by state and animal type;
- Volatile solids (VS) production rate (by animal type and state or U.S.);
- CH4 producing potential (Bo) of the volatile solids (by animal type);
- Methane Conversion Factors (MCF), the extent to which the CH4 producing potential is realized for each type of WMS (by state and manure management system, including the impacts of any biogas collection efforts).

Seven livestock types are considered: dairy cattle, beef cattle, swine, sheep, goats, poultry, and horses. For swine and dairy cattle, manure management system usage is determined for different farm size categories using data from the USDA (USDA 2000a, 2000b, 2000c, 1998b, 1996) and EPA (EPA 2002a, 2002b, ERG 2008, 2000). For beef cattle and poultry, manure management system usage is not tied to farm size and is based on other sources (ERG 2008, 2000, USDA 2000d, UEP 1999). For other animal types, manure management system usage is based on previous estimates (EPA 1992).

Appendix Table A-16 presents a summary of the waste characteristics used in the emissions estimates. The method for calculating volatile solids production from beef and dairy cows, heifers, and steers is based on the relationship between animal diet and energy utilization, which is modeled in the enteric fermentation portion of the inventory. Volatile solids content of manure equals the fraction of the diet consumed by cattle that is not digested and thus excreted as fecal material which, when combined with urinary excretions, constitutes manure. Estimations of gross energy intake and digestible energy were used to calculate the indigestible energy per animal unit as gross energy minus digestible energy plus an additional 2% of gross energy for urinary energy excretion per animal unit. This was then converted to volatile solids production per animal unit using the typical conversion of dietary gross energy to dry organic matter of 20.1 MJ/kg (Garrett & Johnson 1983). Appendix Table A-17 shows volatile solid production rates by state and livestock category.

Methane conversion factors for dry manure management systems were set equal to the default IPCC factors for temperate climates (IPCC 2006). MCFs for liquid slurry, anaerobic lagoon, and deep pit systems were calculated based on the forecast performance of biological systems relative to temperature changes. These calculations account for the following: average monthly ambient temperature, minimum system temperature, the carryover of volatile solids from month to month, and a factor to account for management and design practices that result in loss of volatile solids form lagoon systems. State-level emissions factors for liquid slurry, deep pit, and anaerobic lagoon are shown in Appendix Table A-18. Appendix Table A-19 has national-scale emission factors for other waste management systems. For each animal type, the base emission factors were weighted to incorporate the distribution of waste management systems within each state to get a state-level weighted emission factor (Appendix Table A-20).

Methane emissions were estimated by multiplying regional or national animal type-specific volatile solid production by the animal type-specific maximum CH_4 production capacity of the waste and the state-specific MCF.

The following inputs were used in the calculation of direct and indirect N_2O emissions:

- Animal population data (by animal type and state);
- TAM data (by animal type);
- Portion of manure managed in each WMS (by state and animal type);
- Total Kjeldahl N excretion rate (Nex);
- Direct N_2O emission factor (EFWMS);
- Indirect N_2O emission factor for volatilization (EFvolitalization);
- Indirect N_2O emission factor for runoff and leaching (EFrunoff/leach);
- Fraction of N loss from volatilization of ammonia and NOx (Fracgas);
- Fraction of N loss from runoff and leaching (Fracrunoff/leach)

N_2O emissions were estimated by first determining activity data, including animal population, typical animal mass (TAM), WMS usage, and waste characteristics.

N_2O emissions factors for all manure management systems were set equal to the default IPCC (2006) factors for temperate climates (Appendix A-19).

- Nex rates for all cattle except for bull and calves were calculated for each state and animal type in the Cattle Enteric Fermentation Model (CEFM), which is described in section 6.1, Enteric Fermentation and in more detail in Annex 3.9, Methodology for Estimating CH4 Emissions from Enteric Fermentation. Nex rates for all other animals were determined using data from USDA's Agricultural Waste Management Field Handbook (USDA 1996) and data from the American Society of Agricultural Engineers, Standard D384.1 (ASAE 2003).

- All N_2O emissions factors (direct and indirect) were taken from IPCC (IPCC 2006).
- Country-specific estimates were developed for the fraction of N loss from volatilization (Fracgas) and runoff and leaching (Fracrunoff/leach). Fracgas values were based on WMS-specific volatilization values as estimated from U.S. EPA's *National Emission Inventory - Ammonia Emissions from Animal Agriculture Operations* (EPA 2005). Fracrunoff/leaching values were based on regional cattle runoff data from EPA's Office of Water (EPA 2002b; see Table A-9 in Annex 3.1).

To estimate N_2O emissions, first, the amount of N excreted (kg per year) in manure in each WMS for each animal type, state, and year was calculated. The population (head) for each state and animal was multiplied by TAM (kg animal mass per head) divided by 1,000, the N excretion rate (Nex, in kg N per 1000 kg animal mass per day), WMS distribution (percent), and the number of days per year.

Direct N_2O emissions were calculated by multiplying the amount of Nex (kg per year) in each WMS by the N_2O direct emission factor for that WMS (EFWMS, in kg N2O-N per kg N) and the conversion factor of N_2O-N to N_2O. These emissions were summed over state, animal and WMS to determine the total direct N_2O emissions (kg of N_2O per year).

Then, indirect N_2O emissions from volatilization (kg N_2O per year) were calculated by multiplying the amount of N excreted (kg per year) in each WMS by the fraction of N lost through volatilization (Fractas) divided by 100, and the emission factor for volatilization (EFvolatilization in kg N_2O per kg N), and the conversion factor of N_2O-N to N_2O. Next, indirect N_2O emissions from runoff and leaching (kg N_2O per year) were calculated by multiplying the amount of N excreted (kg per year) in each WMS by the fraction of N lost through runoff and leaching (Fracrunoff/leach) divided by 100, and the emission factor for runoff and leaching (EFrunoff/leach in kg N_2O per kg N), and the conversion factor of N_2O-N to N_2O. The indirect N_2O emissions from volatilization and runoff and leaching were summed to determine the total indirect N_2O emissions.

2.5.2 Uncertainty in Estimating Methane and Nitrous Oxide Emissions from Managed Livestock Waste

The following discussion of uncertainty in estimating GHG emissions from livestock waste is modified from information provided in the U.S. GHG Inventory (EPA 2010, 2007, 2003). The information is reproduced here with permission from EPA.

An uncertainty analysis based on the Monte Carlo Stochastic Simulation technique was conducted on the manure management inventory considering the issues described below and based on published data from scientific and statistical literature, the IPCC, and experts in the industry. The results of the uncertainty analysis showed that the manure management CH_4 inventory has a 95% confidence interval from 39 to 57 Tg CO_2 eq. around the inventory value of 48 Tg CO_2 eq., and the manure management N_2O inventory has a 95% confidence interval from 12 to 18 Tg CO_2 eq. around the inventory value of 14 Tg CO_2 eq (Table 2-1).

Uncertainties derive from limited information on regional patterns in the use of manure management systems and CH_4-generating characteristics of each system. It is assumed that shifts in the swine and dairy sectors toward larger farms causes more manure to be managed in liquid manure management systems. Farm-size data from 1992, 1997 and 2002 are used to modify MCFs based on this assumption. However, the assumption of a direct relationship between farm size and liquid system usage may not apply in all cases and may vary based on geographic location. In addition, the CH_4-generating characteristics of manure management systems are based on relatively few laboratory and field measurements. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC 2000) published a default range of MCFs for anaerobic lagoon systems of 0% to 100%, reflecting the wide range in performance of these systems globally.

There are potential classification errors when naming manure management systems. For example, many livestock waste treatment systems classified as anaerobic lagoons are actually holding ponds, which may be organically overloaded, thus producing CH_4 at a different rate than estimated. In addition, the performance of manure management systems depends on how they are operated, which undoubtedly varies across facilities. An MCF based on optimized lagoon systems does not take into consideration the actual variation in performance across operational systems. Therefore, an MCF methodology was developed to better match observed system performance and account for the impact of temperature on system performance. The MCF methodology used in the inventory includes a factor to account for management and design practices that result in the loss of volatile solids from the management system. This factor, estimated with data from three systems, all in anaerobic lagoons in temperate climates, was applied broadly to systems across a range of management practices. Additional data are needed on animal waste lagoon systems across the country to verify and refine this methodology. Data are also needed on how lagoon temperatures relate to ambient air temperatures and whether the lower bound estimate of temperature used for lagoons and other liquid systems should be revised. The inventory relies on the IPCC MCF for poultry waste management operations of 1.5%. This factor needs further evaluation to assess if poultry high-rise houses promote sufficient aerobic conditions to warrant a lower MCF.

The default N_2O emission factors published in Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC 2000) were derived using limited information. The IPCC factors are global averages; U.S.-specific emission factors may be significantly different. Manure and urine in anaerobic lagoons and liquid/slurry management systems produce CH_4 at different rates, and would in all likelihood produce N_2O at different rates, although a single N_2O emission factor was used for both system types. In addition, there are little data available to determine the extent to which nitrification and denitrification occur in animal waste management systems. Ammonia concentrations that are present in poultry and swine systems suggest that N_2O emissions from these systems may be lower than predicted by the IPCC default factors. At this time, there are insufficient data available to develop U.S.-specific N_2O emission factors; however, this is an area of ongoing research, and warrants further study as more data become available. Similar approaches will be studied for other animal sub-groups.

Additional data would help confirm and track diet changes over time, which are used to introduce variability in volatile solids for beef and dairy cows, heifers, and steers. A similar approach for swine volatile solids production may improve the accuracy of future inventory estimates. Uncertainty also exists with the maximum CH_4-producing potential of volatile solids excreted by different animal groups. The maximum CH_4-producing values used in the CH_4 calculations are published values for U.S. animal waste. However, there are several studies that provide a range of maximum CH_4-producing values for certain animals, including dairy and swine. The maximum CH_4-producing values chosen for dairy assign separate values for dairy cows and dairy heifers to better represent the feeding regimens of these animal groups. For example, dairy heifers do not receive an abundance of high-energy feed and, consequently, their waste will not produce as much CH_4 as would that from milking cows.

2.5.3 Changes Compared to the 2nd Edition of the USDA GHG Report

There were several changes in the methods used to calculate emissions from managed livestock waste. One of the biggest changes is that the Inventory now includes indirect N_2O emissions in the manure management sector associated with N losses from volatilization of N as ammonia (NH_3), nitrogen oxides (NOx), and leaching and runoff, as recommended by IPCC (2006). These indirect N_2O emissions are added to the direct N_2O emissions to present a more complete picture of N_2O emissions from manure management. The days per year used in N_2O calculations was changed from 365 to 365.25 to include leap years and to be consistent with the CH_4 inventory calculations. Instead of calculating state weighted average N_2O emission factors and methane conversion factors (MCFs), N_2O and CH_4 emissions are now calculated from the "bottom up" such that CH_4 and N_2O are calculated for each animal group, manure management system, and state. These values are then summed to calculate the total greenhouse gas emissions from manure management in the United States. Animal population data were updated to reflect the final estimates reports from USDA NASS (USDA 1994, 1998a-b, 2000a, 2004, 2005, 2006, 2007). The FAO (2007) horse population estimates for recent years increased dramatically between the current and previous Inventories, resulting in a much larger estimated horse population, and therefore greater greenhouse gas emissions from this source category. On average, annual CH_4 emission estimates are more than those of the previous Inventory by about one percent. Nitrous oxide emission estimates from manure management systems have increased by approximately 60 percent for all years of the current Inventory compared to the previous Inventory mainly due to accounting both direct and indirect N_2O emissions. The most significant changes in N_2O emissions compared to the previous Inventory occurred in the poultry and swine sectors, whose emissions were approximately 70 percent higher due to the inclusion of indirect N_2O emissions.

2.6 Grazed Lands

For the purposes of this report, the term "grazed lands" refers to all lands grazed by livestock regardless of management intensity (ie., rangeland, pasture, paddock, etc.). Grazed land soils emit N_2O due to enhanced nitrogen cycling as well as a relatively small amount of CH_4 emissions from manure deposits. Manure deposited on grazed land (i.e., unmanaged manure) produces little CH_4 due to predominant aerobic conditions. Nitrous oxide sources include direct and indirect emissions of N_2O associated with increased nitrogen from forage legumes and waste from grazing animals. Grazed lands can be either a source or a sink of CO_2, depending on the level of soil disturbance and grazing intensity but generally sequester carbon because these lands are not plowed.

Nitrous oxide was the predominant GHG emitted from grazed land soils in 2008, accounting for 96% of all emissions from this source (Table 2-6). The remaining 4% of GHG emissions from grazed lands was CH_4. Grazed lands served as a CO_2 sink in 2008, with an uptake of 31.4 Tg CO_2 eq. via the sequestration of CO_2 into soil organic carbon. Nitrous oxide emissions from grazed land totaled 60.5 Tg CO_2 eq. in 2008 (Table 2-6), including direct and indirect sources. Beef cattle are responsible for the highest proportion of direct N_2O emissions from grazed lands because the vast majority of grazed lands in the U.S. are used for beef production. Texas and Oklahoma had the largest emissions from grazed lands due to the large amounts of rangeland in these states. In aggregate, emissions from grazed lands were roughly four times those of managed manure in 2008 and have been since 1990, when national emissions from this source were first estimated (Tables 2-5, 2-6). This is due to large numbers of beef cattle on grazing land (more than 80% of all cattle) compared to feedlots, which are a source of managed waste (Map 2-4).

Table 2-6 Greenhouse Gas Emissions from Grazed Lands in 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
GHG Type						*Tg CO_2 eq.*					
Nitrous Oxide[1]	**64.0**	**62.9**	**64.2**	**56.8**	**58.2**	**63.9**	**64.2**	**59.1**	**60.1**	**61.8**	**60.5**
Direct	53.7	53.3	54.5	49.3	50.2	54.1	54.5	49.6	51.2	52.6	51.3
Indirect Volatilization	5.6	5.5	5.6	5.4	5.1	5.3	5.2	5.3	5.3	5.3	5.3
Indirect Leaching & Run-Off	4.8	4.1	4.2	2.0	2.9	4.5	4.4	4.2	3.6	4.0	3.9
Methane[2]	**2.9**	**3.0**	**2.7**	**2.8**	**2.7**	**2.7**	**2.8**	**2.9**	**3.0**	**3.0**	**2.9**
Carbon Dioxide	**(69.0)**	**(58.9)**	**(83.4)**	**(57.7)**	**(71.4)**	**(31.2)**	**(31.2)**	**(31.3)**	**(31.3)**	**(31.4)**	**(31.4)**
Grazed Lands Remaining Grazed	(46.7)	(36.4)	(51.4)	(27.5)	(43.1)	(4.5)	(4.5)	(4.6)	(4.6)	(4.7)	(4.7)
Land Converted to Grazed Land	(22.3)	(22.5)	(32.0)	(30.2)	(28.3)	(26.7)	(26.7)	(26.7)	(26.7)	(26.7)	(26.7)
Total	**(2.1)**	**6.9**	**(16.4)**	**1.8**	**(10.5)**	**35.4**	**35.7**	**30.7**	**31.8**	**33.4**	**31.9**

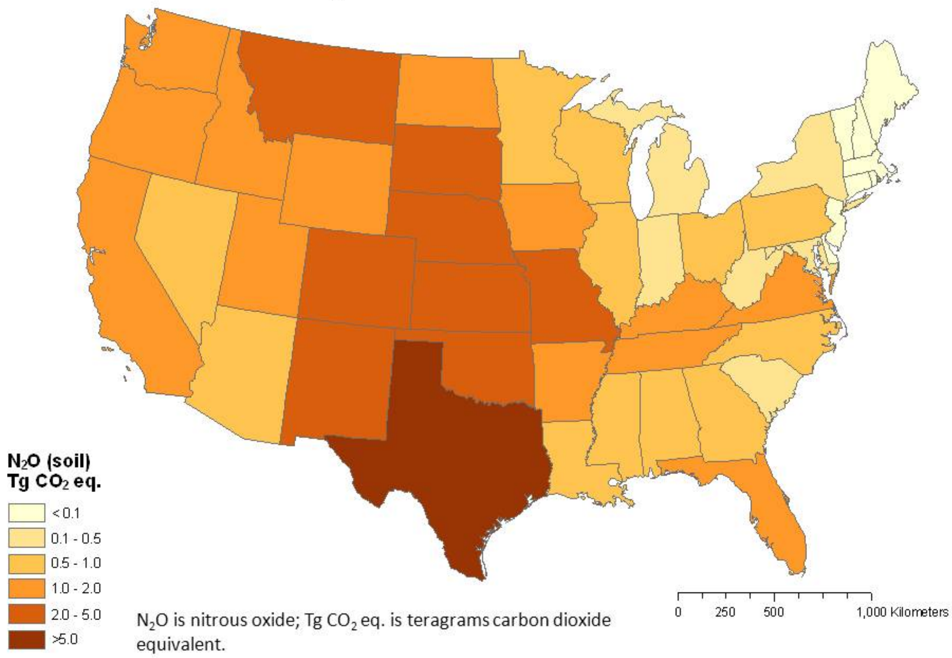
[1] Does not include emissions from managed manure applied to cropped soils.
Tg CO_2 eq. is teragrams carbon dioxide equivalent.

2.6.1 Methodology to Estimate Nitrous Oxide Emissions from Grazed Lands

Estimates of N_2O emissions from this component were based on DAYCENT model simulations of non-federal grazed lands, estimates of animal waste production (Appendix Table A-21), and IPCC (2006) methodology for emissions from federal grazed lands (EPA 2010). Both managed manure applications and unmanaged manure are considered here. Managed manure is defined as manure that was transported and temporarily stored in a management system before soil application. Unmanaged manure is not managed in manure management systems, but instead remains on soils after being deposited by grazing animals in pastures, rangelands, and paddocks.

The livestock included in this component were dairy cattle, beef cattle, swine, sheep, goats, poultry, and horses.

Map 2-4
Nitrous oxide emissions from grazed soils in 2008.



The DAYCENT ecosystem model simulated improved pastures and rangelands at county-level resolution for non-federal grasslands. Improved pastures are defined as grazing lands that have either been seeded with legumes and/or amended with organic nitrogen (e.g., managed manure) or synthetic fertilizer nitrogen. Grazing intensity on improved pastures was assumed to be moderate to heavy, while intensity on rangelands was assumed to be light to moderate. Key model inputs are daily weather, soil texture class, vegetation mix, animal waste N inputs, and grazing intensity. The model simulates soil water and temperature flows, plant growth and senescence, decomposition of dead plant material and soil organic matter, mineralization of nutrients, and trace gas fluxes. Nitrous oxide emissions, nitrate (NO_3) leaching, nitrogen volatilization, animal waste deposition, and nitrogen fixation by legumes were simulated on a per unit area basis, and multiplied by the estimated grazed area (NRI USDA 2000b) in each county to obtain total county level nitrogen losses. The DAYCENT simulations are described in more detail in Chapter 3 of this report and in EPA (2010) and Del Grosso et al. (2006). Manure N deposition from grazing animals (i.e., pasture, range and paddock or PRP manure) was an input to the DAYCENT model (see Annex 3.10 EPA 2010), and included approximately 91 percent of total PRP manure. The remainder of the PRP manure N excretions in each county was assumed to be excreted on federal grasslands, and the N_2O emissions were estimated using the IPCC (2006) Tier 1 method with IPCC default emission factors. Waste nitrogen deposited on grazed lands not accounted for by the DAYCENT simulations were multiplied by the default IPCC (2006) emission factor of 0.02 kg N_20-N/kg N to estimate direct N_2O-nitrogen emissions, as opposed to the 0.01 kg N_2O-N/kg N used to estimate N additions from managed soils (including mineral fertilizers, organic amendments, crop residues, and N mineralization from soil carbon losses).

The amounts of PRP manure N applied on non-federal and federal grasslands in each county were based on the proportion of non-federal grassland area according to data from the NRI

(USDA 2000b, relative to the area of federal grasslands from the National Land Cover Dataset (Vogelman et al. 2001). Sewage sludge was assumed to be applied on grasslands because of the heavy metal content and other pollutants in human waste that limit its use as an amendment to croplands. Sewage sludge application was estimated from data compiled by EPA (2003), McFarland (2001), and NEBRA (2007).

Indirect N_2O emissions due to volatilization of applied nitrogen and indirect N_2O emissions due to leaching were calculated using DAYCENT and IPCC (2006) estimates of volatilization and NO_3 leaching and IPCC estimates of the portion of volatilized or leached/runoff nitrogen that is converted to N_2O. Nitrogen volatilized, leached, or runoff N are all outputs for the grazed lands simulated by DAYCENT. For animal waste not accounted for by the DAYCENT simulations, 10% of animal waste nitrogen was assumed to volatilize and 30% of animal waste nitrogen was assumed to be leached or runoff. The total volatilized nitrogen was multiplied by the IPCC default emission factor of 0.01 kg N_20- N/kg N (IPCC 2006). The total nitrogen leached or runoff was multiplied by the IPCC (2006) default emission factor of 0.0075 kg N_20-N/kg N.

Total grazed land N_2O emissions were partitioned among different animal types by assuming that emissions are linearly proportional to waste nitrogen production.

2.6.2 Uncertainty in Nitrous Oxide Emissions for Grazed Lands

Uncertainty due to model inputs and model structure were quantified. Model inputs used to represent weather, N inputs, and soil texture are not known precisely, and each of these has an associated range of uncertainty represented by a probability density function. Model structural uncertainty refers to the errors inherent in the model. That is, the model is not expected to yield perfect results even if model inputs were precisely known. Combining uncertainties related to model input and model structure yields uncertainty ranges for N_2O in grazed lands that are larger than those reported in the previous Inventory. To address uncertainty in model inputs, a series of Monte Carlo simulations were performed. To address model structural uncertainty, DAYCENT-simulated N_2O emissions were compared with measured emissions from over 10 grassland experiments in North America. IPCC (2006) methodology was used to estimate uncertainties for federal grazed lands not accounted for by the DAYCENT simulations. Uncertainty from the DAYCENT simulated grazed land was combined with uncertainty for remaining grazed lands calculated using IPCC (2006) methodology by using simple error propagation. The calculated 95% confidence interval around the estimate of 62 Tg CO_2 eq. for grazed soil N_2O emissions was 39 to 156 TgCO_2 eq (Table 2-1). Uncertainty calculations are described in detail in Chapter 3 of this report.

2.6.3 Methodology to Estimate Methane Emissions from Grazed Lands

Methane emissions were estimated by multiplying regional or national animal type-specific volatile solid production by the animal type-specific maximum CH_4-production capacity of the waste and the national MCF for manure deposited on grazed lands.

2.6.4 Changes Compared to the 2nd Edition of the USDA GHG Report

In accordance with 2006 IPCC suggested protocol, the N_2O emission factor for grazed land manure associated with horses, sheep, and goats was set to 1 percent. Previously, the 2% emission factor for cattle, swine, and poultry was applied to all livestock categories. In the previous edition, internal calculations in the DAYCENT model were used to derive N additions from unmanaged animal waste. In this edition, these N additions are based on animal population data. This improvement ensures that the data on PRP manure N in the DAYCENT model simulations is consistent with N excretion data from the Managed Livestock Waste section of this Inventory. Another important change relates to indirect emissions form NO_3 leaching. Nitrate leaching was assumed to be an insignificant source of indirect N_2O in grassland systems where the amount of precipitation plus irrigation did not exceed the potential evapotranspiration, as recommended by IPCC (2006). These areas are typically semi-arid to arid, and nitrate leaching to groundwater is a relatively uncommon event. Adopting this recommendation reduced indirect N_2O emissions significantly. In aggregate, these changes resulted in an approximately 40-percent decrease in N_2O emissions from grazed lands on average, primarily due to the new operational version of DAYCENT, revised N additions from grazing animal waste, and reduced impact of NO_3 leaching on indirect N_2O emissions in arid and semi-arid regions.

2.6.5 Methodology to Estimate Carbon Dioxide Fluxes for Grazed Lands

As with N_2O emissions, carbon dioxide (CO_2) fluxes for grasslands were estimated using results from an ecosystem model (CENTURY) and IPCC (2006) methodology. CENTURY (Parton et al. 1994) uses monthly weather data, surface soil texture class, and current and historical vegetation type and land management information to simulate plant growth and senescence, decomposition of dead plant material and soil organic matter, soil water content and temperature, and other ecosystem variables. CENTURY has been parameterized to simulate continuous grasslands and croplands converted to grasslands but not other land uses converted to grasslands. Consequently, IPCC (2006) methodology was used to estimate CO_2 fluxes for land converted from non-agricultural uses to grazed land. Also, CENTURY has not been well tested with organic soils, so IPCC (2006) methodology was also used for grazed organic soils.

Both CENTURY and IPCC (2006) methodologies rely on land use classifications and land use histories. The National Resources Inventory (NRI USDA 2000b) was used to identify grassland remaining grassland and land converted to grassland. Grassland includes pasture and rangeland where the primary land use is livestock grazing. The NRI is a statistically based sample of all non-federal land and includes ~400,000 points in agricultural land. Data have been reported every five years starting in 1982, and 2003 is the most recent year that has been reported. According to NRI data, ~17 million ha of grassland (out of a total ~261 million ha reported in 2003) were converted to grassland between 1997 and 2003. An example of land converted to grassland is land that was cropped historically but then converted to pasture use. Carbon dioxide fluxes for grazed lands were calculated using estimates of changes in soil organic carbon stocks and molecular stoichiometry.

Mineral soil carbon stocks and stock changes for NRI points classified as grasslands remaining grasslands and cropland converted to grassland were estimated using the CENTURY model. In addition to accounting for weather and soil texture, these simulations also included estimates of managed manure additions to grasslands. Waste from grazing animals deposited directly onto grasslands is calculated by the model based on grazing intensity and forage availability. CENTURY estimates carbon stock changes by accounting for carbon inputs from plant material and manure and carbon outputs from grazing and decomposition. For details on sources of the input data required to run CENTURY and how the simulations were conducted, see Chapter 3 of this report and Chapter 7 and Annex 3.13 of the U.S. GHG Inventory (EPA 2010).

Mineral soil carbon stocks and stock changes for NRI points classified as land other than cropland converted to grassland and all grasslands growing on organic soils were estimated using IPCC (2006, 1997) methodology. U.S.-specific stock change factors based on field data were developed for land converted to grassland and for drained histosols used for grazing. As with grazed land N_2O emissions, CO_2 fluxes were partitioned among different animal types by assuming that fluxes are linearly proportional to waste nitrogen production.

2.6.6 Uncertainty in Carbon Dioxide Fluxes for Grazed Lands

Uncertainty for the estimates of CO_2 fluxes from mineral soil grassland remaining grassland and cropland converted to grassland provided by CENTURY model simulations used a Monte Carlo approach, which addresses uncertainties in model inputs and uncertainties from scaling NRI points to cover all grasslands remaining grassland in the U.S. Uncertainty for estimates from other land uses converted to grassland and all organic soil grasslands provided by IPCC (2006, 1997) methodology used a Monte Carlo approach that addressed uncertainties in carbon stock change factors and in land use data. Uncertainties were combined using simple error propagation, the results yielded an uncertainty of (7) to (3) around the estimate of (5) Tg CO_2 eq. in 2008 for land remaining grazed land and (29) to (24) around the estimate of (27) Tg CO_2 eq. for land converted to grazed land in 2008, where parentheses indicate a net sequestration of CO_2 (Table 2-1).

2.6.7 Changes Compared to the 2nd Edition of the USDA GHG Report

There are several important changes that impacted estimate of carbon dioxide fluxes for grazed lands. Annual survey data from the USDA National Resources Inventory (NRI) were incorporated into this year's Inventory. This resulted in the availability of new data, which extended the time series of activity data beyond 1997 to 2003. In previous Inventories, activity data were only available through 1997 at 5-year intervals, and subsequent years were treated as the same land use practice occurring in 1997. Each NRI point was simulated separately, instead of simulating clusters of points that had common land use histories and soil characteristics in a county as was done previously. NRI area data were reconciled with the forest area estimates in the Forest Inventory and Analysis (FIA) dataset, and were incorporated into the estimation of soil C stock changes. Overall, these changes resulted in an average annual increase in soil C stocks of approximately 40 Tg CO_2 eq. for the time series, compared to the previous Inventory.

2.7 Mitigating Greenhouse Gas Emissions from Livestock

2.7.1 Enteric Fermentation

Emissions of CH_4 from enteric fermentation in ruminant and non-ruminant animals are dependent on the animal's digestive system and the amount and type of feed consumed. On average, beef and dairy cattle convert 6% of gross energy intake from feed into CH_4 through enteric fermentation, constituting a loss of energy from the perspective of the animal (Johnson & Johnson 1995). Research on animal nutrition has focused on reducing this energy loss, which consequently reduces CH_4 emissions and increases nutritional efficiency. Through such research, a number of potential strategies have been identified to reduce CH_4 emissions from enteric fermentation, including (Mosier et al. 1998):

- Increasing the digestibility of forages and feeds;
- Providing feed additives which may tie up hydrogen in the rumen;
- Inhibiting the formation of CH4 by rumen bacteria;
- Increasing acetic acid in the rumen;
- Improving production efficiency; and
- Modifying bacteria in the rumen.

Currently, government research programs indirectly address mitigation of CH_4 emissions through improved livestock production. Ongoing research development and deployment efforts related to mitigating CH_4 emissions include:

- Decreasing feed digestion time by improving grazing management to increase the digestibility of forages, increasing the digestibility of feed grains, and increasing the feeding of concentrated supplements;
- Adding edible oils in feed to sequester hydrogen making it unavailable for methanogens;
- Using feed additives, ionophores, which inhibit the formation of CH_4 by rumen bacteria;
- Improving livestock production efficiency by feed additives such as hormones to increase milk production and growth regulators for beef production or by improved diet or genetics;
- Enhancing rumen microbes to produce usable products rather than CH_4.

2.7.2 Livestock Waste

Livestock and poultry waste from production facilities has the potential to produce significant quantities of CH_4 and N_2O, depending on the waste management practices used. In the United States, livestock and poultry manure is managed in a myriad of ways, suggesting there are multiple options for reducing CH_4 and N_2O emissions. When manure is stored or treated in systems that promote anaerobic conditions, such as lagoons and tanks, the decomposition of the biodegradable fraction of the waste tends to produce CH_4. When manure is handled as a solid, such as in stacks or deposits on pastures, the biodegradable fraction tends to decompose aerobically and produce little or no CH_4, although it produces N_2O.

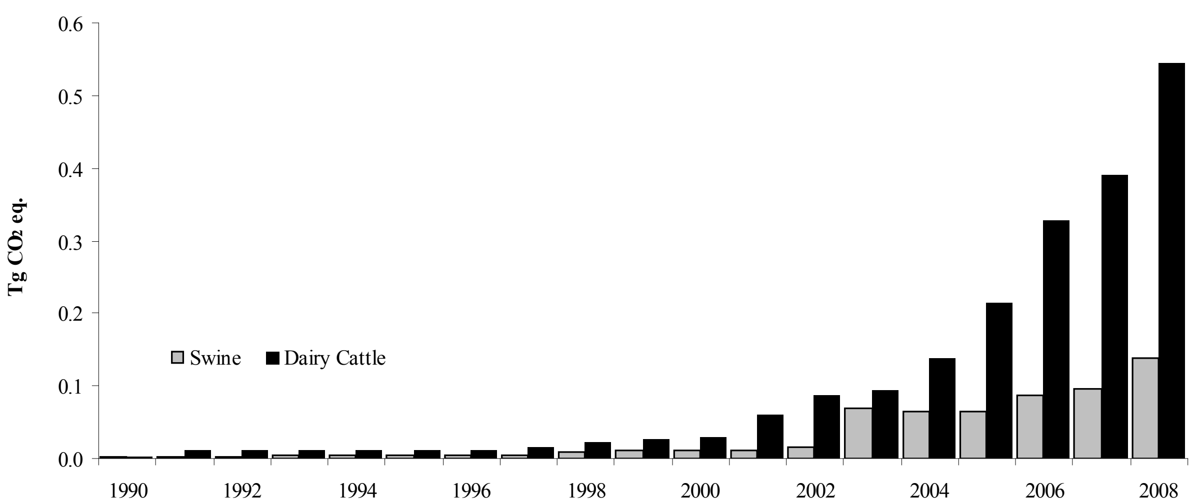
A relatively large portion of CH_4 is emitted from livestock and poultry waste in anaerobic lagoons. Current, commercially available technologies that have been the most successful in reducing CH_4 emissions from manure management are anaerobic digestion systems. Unlike conventional lagoons, digestion technologies keep waste treatment and storage functions separate and allow for gas recovery and combustion, pathogen and organic stabilization, odor and other air quality pollution control, and flexible approaches to nutrient management.

The EPA tracks installation and usage of anaerobic digesters under voluntary programs such as AgStar (http://www.epa.gov/agstar/) and uses this data to estimate how much anaerobic digesters have reduced overall CH_4 emissions from livestock waste over the last 11 years. Figure 2-4 shows an increasing trend in emissions reductions annually from the use of anaerobic digesters, reflecting increasing numbers of digester systems being installed each year.

Other emission reduction processes can include separation, aeration, or shifts to solid handling or storage management systems. These strategies, however, could be limited by other farm or environmental constraints and costs.

Figure 2-4

Estimated Reductions in Methane Emissions from Anaerobic Digesters, 1990-2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent

2.7.3 Grazed Lands

Nitrous oxide is by far the largest source of emissions from grazed lands so it also provides the largest mitigation potential (Table 2-6). However, because grazed lands are not highly managed, particularly the large expanses of rangeland in the Western U.S., mitigation options are limited.

One strategy that may be feasible for more intensely managed pastures in the Eastern U.S. is nitrification inhibitors. Although synthetic nitrogen fertilizer inputs are low, grazing lands usually have large nitrogen inputs from biological nitrogen fixation because they are seeded with legumes. This mitigation potential has not been quantified but will be in future DAYCENT model simulations.

Recent model simulations indicate grazed lands are currently providing a net sequestration of CO_2 emissions (Table 2-6) and have the potential to store over 100 Tg CO_2 per year across the U.S. (Follett et al. 2001). The largest potential is by decreasing soil erosion and restoring eroded and degraded soils so that they become net carbon sinks. Other management practices which enhance carbon storage include nutrient/manure additions, legume seeding, and improved grazing management. However, the benefits of increased carbon storage must be compared with the costs of increased N_2O emissions associated with nutrient/manure additions and legume seeding.

Chapter 3: Cropland Agriculture

3.1 Summary of U.S. Greenhouse Gas Emissions from Cropland Agriculture

In 2008, cropland agriculture resulted in total emissions of 196 Tg CO_2 eq. of greenhouse gases (GHG) (Table 3-1). Cropland agriculture is responsible for almost half (46%) of all emissions from the agricultural sector (EPA 2010). Nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4) emissions from cropped soils totaled 154, 34, and 8 Tg CO_2 eq., respectively, in 2008. However, that amount was offset by a storage, or carbon sequestration, of 42 Tg CO_2 eq. in cropped soils in 2008. When carbon sequestration is taken into account, net emissions of GHG from cropland agriculture amount to approximately 154 Tg CO_2 eq. The 95% confidence interval for net emissions in 2008 is estimated to lie between 104 and 246 Tg CO_2 eq. (Table 3-1).

Table 3-1 Estimates and Uncertainties for Cropland Greenhouse Gas Emissions, 2008

	GHG Emissions	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Source		*Tg CO_2 eq.*			*percent*
N_2O	**154**	**114**	**241**	**-26**	**+57**
Soils Direct	118	84	181	-29	+53
Soils Indirect[1]	35	14	96	-59	+173
Residue Burning	1	0	1	-71	+83
CH_4	**8**	**4**	**19**	**-57**	**+127**
Residue Burning	1	0	2	-68	+88
Rice Cultivation	7	3	18	-64	+143
CO_2	**(8)**	**(38)**	**20**	**-360**	**+347**
Mineral Soils	(42)	(69)	(16)	-63	+63
Organic Soils	30	17	40	-43	+33
Liming of Soils	4	0	8	-97	+102
Total Emissions	**196**	**154**	**285**	**-22**	**+45**
Net Emissions[2]	**154**	**104**	**246**	**-33**	**+60**

Note: Parentheses indicate a net sequestration. Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide; CO_2 is carbon dioxide.
[1] Accounts for loss of manure N during transport, treatment and storage, including volatilization and leaching/runoff.
[2] Includes sources and sinks.

Net emissions in 2008 were 23% higher than the baseline year (1990). Greenhouse gas emissions from agricultural soils fluctuated between 1990 and 2008, with CH_4 and N_2O reaching their highest levels in 2001 (Table 3-2). Net CO_2 flux showed substantial interannual variability, mainly due to fluctuations in the mineral soil CO_2 sink. Annual fluctuations in CO_2 sequestration are primarily a result of variability in weather patterns and land use changes.

Table 3-2 Summary of Greenhouse Gas Emissions from Cropland Agriculture, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Source						*Tg CO_2 eq.*					
N_2O	**139.5**	**144.1**	**151.8**	**160.2**	**150.2**	**147.8**	**152.4**	**153.8**	**150.5**	**151.2**	**153.9**
Soils Direct	103.0	109.8	115.6	122.3	115.3	111.4	118.5	117.9	114.7	116.7	118.3
Soils Indirect[1]	36.0	33.9	35.7	37.5	34.4	35.9	33.4	35.4	35.3	34.1	35.1
Residue Burning	0.4	0.4	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
CH_4	**7.9**	**8.4**	**8.4**	**8.5**	**7.6**	**7.8**	**8.5**	**7.8**	**6.8**	**7.1**	**8.2**
Residue Burning	0.8	0.7	0.9	0.9	0.8	0.9	1.0	0.9	0.9	1.0	1.0
Rice Cultivation	7.1	7.6	7.5	7.6	6.8	6.9	7.6	6.8	5.9	6.2	7.2
CO_2	**(22.6)**	**(15.6)**	**(23.5)**	**(4.3)**	**(1.7)**	**(7.2)**	**(8.3)**	**(8.0)**	**(8.9)**	**(9.2)**	**(8.3)**
Mineral Soils	(57.1)	(50.3)	(58.1)	(39.0)	(37.0)	(42.0)	(42.5)	(42.6)	(43.4)	(44.0)	(42.4)
Organic Soils	29.8	30.3	30.3	30.3	30.3	30.3	30.3	30.3	30.3	30.3	30.3
Liming of Soils	4.7	4.4	4.3	4.4	5.0	4.6	3.9	4.3	4.2	4.5	3.8
Total Emissions	**181.9**	**187.2**	**194.8**	**203.4**	**193.1**	**190.5**	**195.2**	**196.2**	**191.9**	**193.2**	**196.2**
Net Emissions[2]	**124.8**	**136.9**	**136.7**	**164.5**	**156.1**	**148.4**	**152.7**	**153.6**	**148.5**	**149.2**	**153.8**

Note: Parentheses indicate a net sequestration. Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide; CO_2 is carbon dioxide.

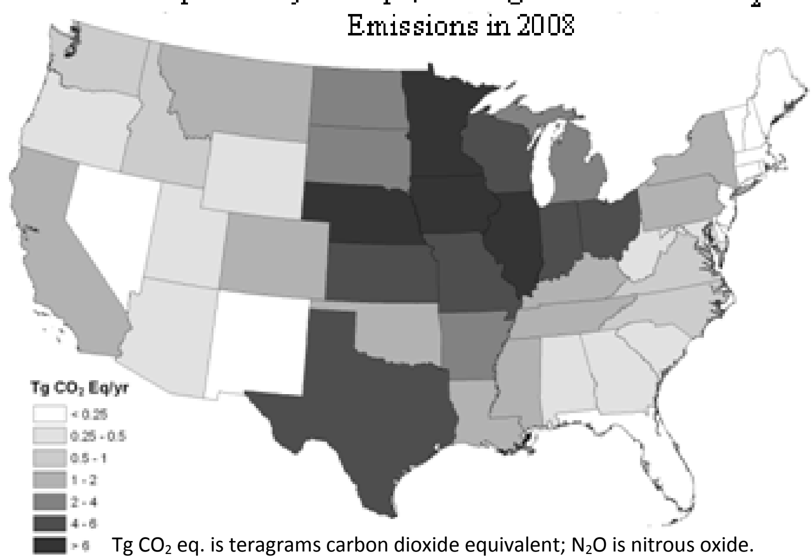
[1] Soils Indirect N_2O emissions account for volatilization and leaching/runoff.

[2] Includes sources and sinks.

Greenhouse gas emission from agricultural soils, primarily N_2O, were responsible for the majority of total emissions, while CH_4 and N_2O from residue burning and rice cultivation caused about 4% of emissions in 2008 (Tables 3-1, 3-2). Soil CO_2 emissions from cultivation of organic soils (15%) and from liming (2%) are the remaining sources. Nitrous oxide emissions from soils are the largest source in the U.S. because N_2O is a potent greenhouse gas (see Chapter 1 Box 1-1) and due to the large amounts of nitrogen added to crops in fertilizer that stimulate N_2O production. Emissions from residue burning are minor because only ~3% of crop residue is assumed to be burned in the U.S. (EPA 2010). Cropped soils in the U.S. are a net CO_2 sink mainly because reduced tillage intensity has become more popular in recent years and lands used for perennial hay cropping, as well as idle cropland enrolled in the Conservation Reserve Program (CRP), continue to store carbon.

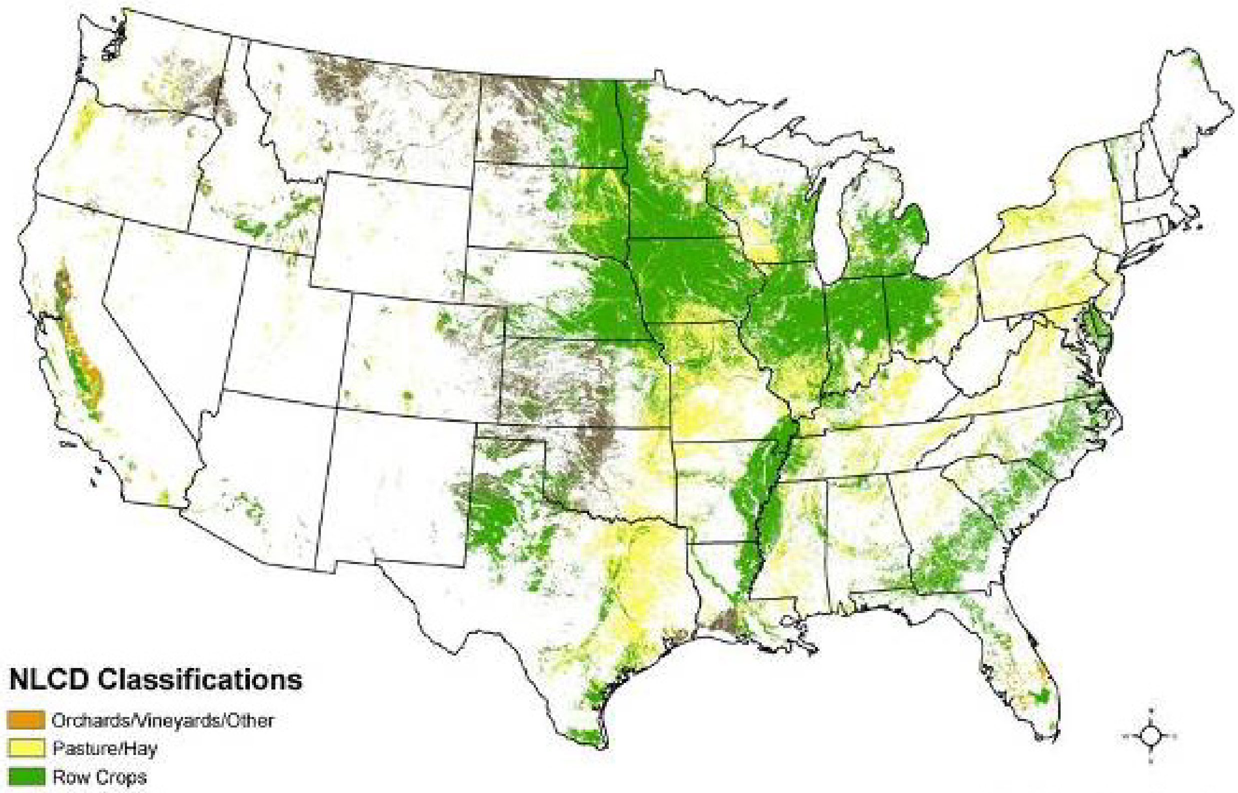
Nitrous oxide emissions were largest in areas where a large portion of land is used for intensive agriculture (Map 3-1). For example, 90% or more of the land in many counties in the Corn Belt is intensively cropped (Map 3-2). Corn is the leading crop for N_2O

Map 3-1 Major Crops, Average Annual Direct N_2O Emissions in 2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent; N_2O is nitrous oxide.

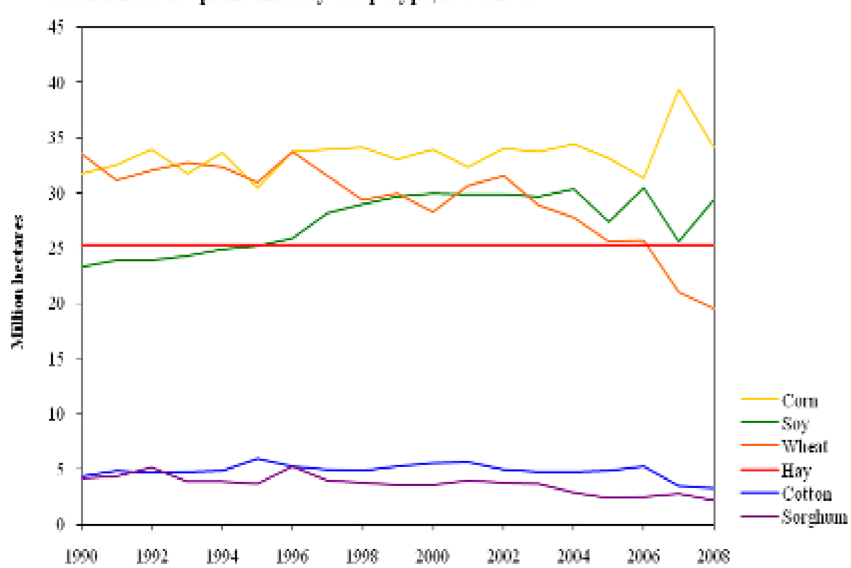
Map 3.2 U.S. Cropped Land



emissions followed by soybean and hay (Table 3-3).

Emissions from corn cropping are high because large amounts of nitrogen (N) fertilizer are routinely applied and the land area used for corn production is the most extensive (Figure 3-1). Although little N fertilizer is applied for soybean cropping, N_2O emissions are high because soybeans supply large amounts of N to the soil from biological fixation of atmospheric nitrogen (N_2). In general, N_2O emissions are highly correlated with crop areas and nitrogen inputs. Biological fixation makes up about half of total N additions, followed by synthetic fertilizer addition and manure (Figure 3-2). Note that Map 3-1 does not include emissions from non-major crops, which make up a significant portion of total emissions in California and Florida. The cropped soil emissions reported here are consistent with those in EPA (2010).

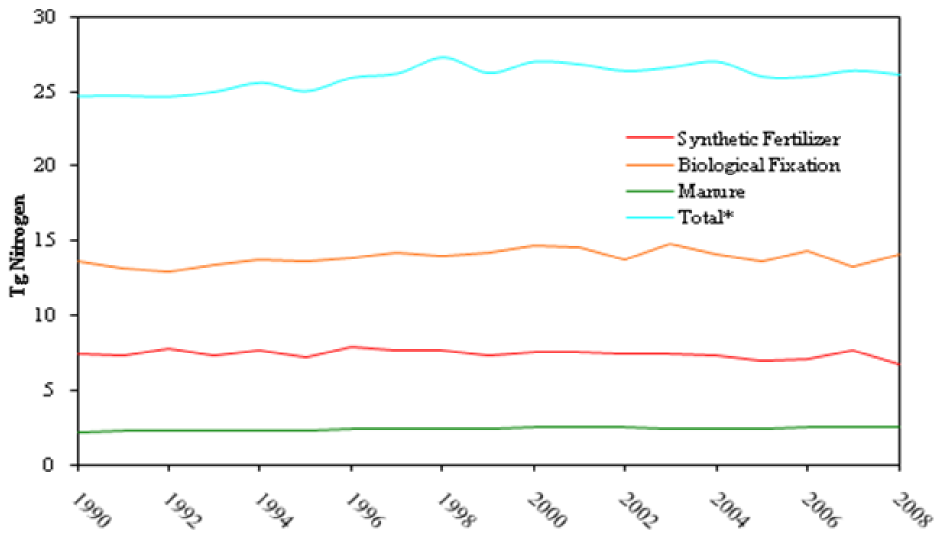
Figure 3-1
U.S. Planted Cropland Area by Crop Type, 1990-2008



Cropland agriculture results in GHG emissions from multiple sources, with the magnitude of emissions determined, in part, by land management practices. Application of synthetic and organic fertilizers, cultivation of N fixing crops and rice, cultivation and management of soils, and field burning of crop residues leads to emissions of N_2O, CH_4, and CO_2. However, agricultural soils can also mitigate GHG emissions through the biological uptake of organic carbon in soils resulting in CO_2 removals from the atmosphere. This chapter covers both GHG emissions from cropland agriculture and biological uptake of CO_2 in agricultural soils. National estimates of these sources, published in the U.S. GHG Inventory, are reported in this section and, where appropriate, county and state-level emissions estimates are provided.

Sources and sinks of N_2O, CH_4, and CO_2 and the mechanisms that control fluxes are discussed in detail. Methodologies used to estimate emissions are summarized and mitigation opportunities are discussed and quantified where possible. The methodologies used here are similar to those reported in the second edition of the USDA GHG report (USDA 2008), with some improvements in model algorithms and model input data.

Figure 3-2
Annual Nitrogen Inputs to Cropland Soil



*Includes activated and other sludge, dried blood, compost, tankage, and other organics.

Table 3-3 Nitrous Oxide Emissions from Differently Cropped Soils, 1990, 1995, 2000-2008[1]

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Source						*Tg CO_2 eq.*					
Corn	**47.5**	**42.8**	**49.7**	**53.6**	**49.3**	**47.8**	**51.6**	**51.6**	**47.1**	**59.3**	**54.0**
Direct	36.1	34.8	40.0	42.8	40.3	37.4	42.5	41.7	38.0	48.0	43.7
Volatilization	1.1	1.1	1.3	1.2	1.3	1.2	1.3	1.2	1.2	1.5	1.3
Leaching & Runoff	10.2	6.9	8.3	9.6	7.7	9.2	7.9	8.7	7.8	9.8	9.0
Soybean	**23.8**	**22.2**	**29.7**	**33.1**	**28.7**	**29.0**	**29.9**	**28.7**	**30.1**	**25.4**	**28.8**
Direct	17.1	17.7	22.5	24.5	22.0	21.2	22.5	21.6	22.8	19.3	21.8
Volatilization	0.9	0.9	1.2	1.1	1.1	1.1	1.2	1.0	1.1	1.0	1.1
Leaching & Runoff	5.8	3.6	5.9	7.4	5.7	6.6	6.2	6.1	6.2	5.2	5.9
Hay	**16.8**	**16.4**	**17.5**	**18.6**	**16.8**	**17.2**	**17.0**	**17.9**	**16.8**	**17.3**	**17.4**
Direct	14.3	13.7	15.4	15.8	14.4	14.6	15.0	15.3	14.7	14.9	15.2
Volatilization	0.3	0.3	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3
Leaching & Runoff	2.2	2.4	1.8	2.4	2.1	2.3	1.7	2.3	1.7	2.0	1.9
Wheat	**13.0**	**17.8**	**10.8**	**10.7**	**11.5**	**11.9**	**9.9**	**8.6**	**10.6**	**8.7**	**8.2**
Direct	10.1	8.8	8.1	8.3	8.7	7.7	7.9	7.1	7.1	6.4	6.3
Volatilization	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.4	0.3
Leaching & Runoff	2.4	8.5	2.3	1.9	2.4	3.8	1.6	1.2	3.1	1.9	1.6
Cotton	**3.9**	**5.5**	**5.9**	**5.8**	**5.4**	**4.5**	**4.6**	**5.3**	**4.7**	**3.6**	**3.4**
Direct	3.2	4.1	4.4	4.7	3.9	3.7	3.6	4.2	4.1	2.8	2.6
Volatilization	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Leaching & Runoff	0.6	1.3	1.4	0.9	1.4	0.7	0.9	1.0	0.5	0.8	0.7
Sorghum	**2.8**	**2.2**	**2.2**	**2.7**	**2.5**	**2.0**	**1.6**	**1.5**	**1.6**	**1.7**	**1.9**
Direct	2.1	1.8	1.8	2.3	2.0	1.6	1.3	1.2	1.2	1.4	1.5
Volatilization	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0
Leaching & Runoff	0.7	0.3	0.3	0.3	0.5	0.3	0.3	0.3	0.4	0.3	0.4
Non-major crops	**18.8**	**23.4**	**22.0**	**21.2**	**21.5**	**21.8**	**23.8**	**26.0**	**25.1**	**20.5**	**25.7**
Direct	14.1	17.6	16.5	16.0	16.3	16.6	18.1	19.7	18.9	15.6	19.5
Volatilization	1.8	2.2	2.2	2.1	2.2	2.1	2.3	2.5	2.5	2.1	2.5
Leaching & Runoff	2.9	3.6	3.3	3.1	3.1	3.1	3.4	3.8	3.7	2.9	3.8
Histosol Cultivation[2]	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**
Managed Manure[3]	**9.9**	**10.8**	**11.2**	**11.6**	**11.6**	**10.6**	**11.1**	**11.3**	**11.6**	**11.7**	**11.6**
All Direct	109.8	112.2	122.8	129.0	121.9	116.4	124.8	125.0	121.3	123.0	125.1
All Volatilization	5.0	5.2	5.7	5.5	5.5	5.4	5.7	5.6	5.7	5.3	5.6
All Leaching & Runoff	24.7	26.6	23.4	25.7	22.9	25.9	21.9	23.3	23.4	22.9	23.2
Total	**139.5**	**144.1**	**151.8**	**160.2**	**150.2**	**147.8**	**152.4**	**153.8**	**150.5**	**151.2**	**153.9**

Note: Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide; CO_2 is carbon dioxide.
[1] Emissions from residue burning are not included.
[2] Direct emissions.
[3] Accounts for loss of manure N during transport, treatment and storage, including volatilization and leaching/runoff.

3.2 Sources of Greenhouse Gas Emissions in Cropland Agriculture

3.2.1 Cropped Soils

Agricultural soils serve as both a source of GHG and a mechanism to remove CO_2 from the atmosphere. Nitrous oxide, CH_4, and CO_2 emissions and sinks are a function of underlying biochemical processes. Nitrous oxide is produced as an intermediate during nitrification and denitrification in soils (Firestone & Davidson 1989). In nitrification, soil micro-organisms ("microbes") convert ammonium (NH_4) to nitrate (NO_3) through aerobic oxidation (IPCC 1996). In denitrification, microbes convert nitrate to nitrogen oxides (NO_x) and dinitrogen gas (N_2) by anaerobic reduction. During nitrification and denitrification, soil microbes release N_2O, which can diffuse from the soil and enter the earth's atmosphere (IPCC 1996). Cropland soil amendments that add nitrogen to soils drive the production of N_2O by providing additional substrate for nitrification and denitrification. Commercial fertilizer, livestock manure, sewage sludge, cultivation of N-fixing crops, and incorporation of crop residues all add N to soils. In addition, cultivation, particularly of soils high in organic matter (i.e., histosols), enhances mineralization of nitrogen-rich organic matter, making more nitrogen available for nitrification and denitrification (EPA 2010, 2007). Compared to soil N_2O emissions, other GHG sources from croplands are relatively small. Methane gas is produced and emitted primarily from rice paddies. This, however, is responsible only for a small portion of total emissions from cropped soils in the U.S. due to the small land area cropped with rice in this country. Emissions from crop residue burning are also not a large source compared to soils due to the small portion of residues burned in the U.S.

Nitrous oxide is the major GHG emitted from cropland agriculture in the U.S. Nitrogen can be converted to N_2O and emitted directly from agricultural fields (direct emissions), or it can be transported from the field in a form other than N_2O and then converted to N_2O elsewhere (indirect emissions). A major source of indirect N_2O emissions is from nitrate that either leaches into the groundwater or runs off the soil surface and then is converted to N_2O via aquatic denitrification (Del Grosso et al. 2006). A second source of indirect N_2O emissions comes from N that is volatilized to the atmosphere, then is deposited back onto soils, and converted to N_2O (Del Grosso et al. 2006).

The size of CO_2 sources and sinks from soils is related to the amount of organic carbon stored in the soil (IPCC 1996). Changes in soil organic carbon (SOC) content are related to inputs (e.g., atmospheric CO_2 fixed as carbon in plants through photosynthesis) and losses from decomposition of soil organic matter which causes CO_2 emissions (IPCC 1996). The net balance of CO_2 uptake and loss in soils is driven in part by biological processes, which are affected by soil characteristics and climate. In addition, land use and management can affect the net balance of CO_2 through modifying inputs and rates of decomposition (IPCC 1996). Changes in agricultural practices such as clearing, drainage, tillage, crop selection, irrigation, grazing, crop residue management, fertilization, and flooding can modify both organic matter inputs and decomposition, and thereby result in a net flux of CO_2 to or from soils.

Most agricultural soils contain comparatively low amounts of organic carbon as a percentage of total soil mass, typically in the range of 1 to 6 % organic C by weight, and are thus classified as mineral soils (NRCS 1999). However, on an area basis, this amount of carbon typically exceeds that stored in vegetation in most ecosystems (including forests). Historically, conversion of native ecosystems to agricultural uses resulted in large soil carbon losses, as much as 30-50% or more of the C present in the native condition (Haas et al. 1957, Schlesinger 1986, Guo & Gifford 2002, Lal 2004). Presently, after many decades of cultivation, most soils have likely stabilized at lower carbon levels or are increasing their organic matter levels as a result of increasing crop productivity (providing more residues), less intensive tillage, and other improvements in agricultural management practices (Paustian et al. 1997, Allmaras et al. 2000, Follett 2001). Changes in land-use or management practices that result in increased organic inputs or decreased oxidation of organic matter (e.g., taking cropland out of production, improved crop rotations, cover crops, application of organic amendments and manure, and reduction or elimination of tillage) usually result in a net accumulation of SOC until a new equilibrium is achieved.

Cultivated organic soils, also referred to as histosols, contain more than 12 to 20% organic matter by weight, and constitute a special case (NRCS 1999, Brady & Weil 1999). Organic soils form as a result of water-logged conditions, in which decomposition of plant residue is retarded. When organic soils are drained and cultivated, the rate of decomposition, and hence CO_2 emissions, is greatly accelerated. Due to the depth and richness of the organic layers, carbon loss from cultivated organic soils can continue over long periods of time.

In addition, lime, often added to mineral and organic agricultural soils to reduce acidic conditions, contains carbonate compounds (e.g., limestone and dolomite) that when added to soils release CO_2 through the bicarbonate equilibrium reaction (IPCC 1996).

3.2.2 Rice Cultivation

Rice cultivation is unique because it takes place almost exclusively on flooded fields, including in the U.S. where rice is almost grown entirely on flooded fields (EPA 2010). This water regime causes CH_4 emissions as a result of waterlogged soils restricting oxygen diffusion and creating conditions for anaerobic decomposition of organic matter, facilitated by CH_4 emitting "methanogenic" bacteria (IPCC 1996, Le Mer & Roger 2001). Methane from rice fields reaches the atmosphere in three ways: bubbling up through the soil, diffusion losses from the water surface, and diffusion through the vascular elements of plants (IPCC 1996). Diffusion through plants is considered the primary pathway, with diffusion losses from surface water being the least important process (IPCC 1996). Soil composition, texture, and temperature are important variables affecting CH_4 emissions from rice cultivation, as are the availability of carbon substrate and other nutrients, soil pH, and partial pressure of CH_4 (IPCC 1996). Since U.S. rice acreage is relatively small compared to other crops, CH_4 emissions from rice cultivation are small compared to other cropland agriculture sources (EPA 2007).

3.2.3 Residue Burning

In the U.S., 7-8 million acres of crop residues in fields are burned annually to prepare for cultivation and to control for pests (EPA 2010). While CO_2 is a product of residue combustion, residue burning is not considered a net source of CO_2 to the atmosphere because CO_2 released from burning crop biomass is replaced by uptake of CO_2 in crops growing the following season (IPCC 1996). However, CH_4 and N_2O, also products of residue combustion, are not recycled into crop biomass through biological uptake the following season. Therefore, residue burning is considered a net source of CH_4 and N_2O to the atmosphere. Overall, GHG emissions from field burning of crop residues are comparatively small in the U.S. relative to other countries (EPA 2010).

3.2.4 Agroforestry

Agroforestry practices such as establishing windbreaks and riparian forest buffers represent another potential carbon sink in cropland agriculture. Comprehensive data on agroforestry practices are not available to estimate the current national levels of carbon sequestration from such practices. However, published research studies have estimated the potential agroforestry carbon sink in the U.S. In temperate systems, agroforestry practices store large amounts of carbon (Kort & Turlock 1999, Schroeder 1994), with the potential ranging from 15 to 198 metric tons of carbon per hectare (modal value of 34 metric tons of carbon per hectare) (Dixon 1995). Nair and Nair (2003) estimated that by the year 2025, the potential carbon sequestration of agroforestry in the United States will be 90 million metric tons of carbon per year. There is a need to better quantify and track agroforestry practices nationally, particularly to inform USDA programs like the Conservation Reserve Program, Environmental Quality Incentives Program, and Forest Land Enhancement Program, which may provide incentives to land owners to implement agroforestry.

3.3 Nitrous Oxide Emissions from Cropped Soils

In 2008, 80% of total cropland soil N_2O emissions were direct soil emissions (Table 3-3). Of the 20% of total emissions from indirect N_2O, 81% are from NO_3 leaching/runoff and the remainder are associated with volatilization. Corn cropland has the highest emissions, roughly 35% of the total, followed by soybean and hay (Table 3-3). Emissions are highest from corn because corn covers the largest land area (Figure 3-1) of all crops and synthetic nitrogen inputs with corn are high. Emissions from soybeans are high due to large crop area and high rates of nitrogen fixation. Other factors contributing to high emissions for these crops are: they are grown mostly in the north central region where many of the soils are high in organic matter and some of the soils are poorly drained, both of which enhance denitrification rates. In the previous report, emissions from wheat were third highest, but recent declines in wheat area have resulted in fewer emissions for this particular crop. Emissions from hay cropping are substantial, despite minimal fertilizer N additions, because a large portion of hay includes N-fixing plants (e.g., alfalfa). Emissions from cotton and sorghum are low, as the cropland area for these crops is small compared to the other major crops simulated by DAYCENT. In addition, emissions from sorghum are low because this crop tends to be grown in drier areas in the eastern Great Plains, and cotton is grown mostly in the South, where soils tend to be low in organic matter. Non-major crop types were responsible for ~17% of total emissions in 2008 (Table 3-3). Emissions from histosol cultivation are small (~2% of total) because histosols represent only ~750,000 ha, which is less than 1% of U.S. cropped land.

Nitrous oxide emissions are largely driven by nitrogen additions, weather, and soil physical properties. External nitrogen inputs (i.e., addition of synthetic fertilizers and manure, as well as biological fixation) to cropped soils varied between ~24 and 27 Tg N between 1990 and 2008 (Fig. 3-2), while N_2O emissions varied between 142 and 165 Tg CO_2 eq. (Table 3-3). Variation in N inputs explained roughly 46% of the variability in soil N_2O emissions. Also, the years with highest nitrogen inputs did not necessarily lead to the highest N_2O emissions. This indicates that other factors such as changes in weather patterns strongly influence the annual variability in estimated N_2O emissions. Specifically, amount and timing of precipitation, temperature patterns, and soil carbon and nitrogen availability interact to control N_2O emissions.

3.3.1 Methods for Estimating N_2O Emissions from Cropped Soils

Emissions of N_2O from nitrogen additions to cropland soils and cultivation of histosol soils are source categories analogous to those covered in Agricultural Soil Management in the U.S. GHG Inventory (EPA 2010), with some exceptions. The U.S. GHG Inventory (EPA 2010) includes in the Agricultural Soils Management section direct emissions of N_2O from livestock on grazed lands, while the USDA GHG Inventory includes this source under Livestock GHG Emissions in Chapter 2 of this report. Also, the U.S. GHG Inventory (EPA 2010) includes in the Agricultural Soils Management section indirect emissions of N_2O from all sources, including indirect N_2O from livestock grazing and from urban areas. For this report, indirect N_2O from grazing is included in the livestock chapter, while indirect emissions from urban areas and other non-agricultural sources are not covered at all.

Briefly, the DAYCENT ecosystem model was used to estimate direct soil N_2O emissions, NO_3 leaching, and nitrogen volatilization from major crop types. IPCC (2006) methodology was used to estimate direct and indirect emissions from cropped soils not included in the DAYCENT simulations and to calculate indirect emissions from DAYCENT estimates of NO_3 leaching and volatilization. IPCC (2006) methodology was also used to estimate emissions from cultivation of organic soils. Use of a process-based model for inventories is known as a Tier 3 approach, while use of IPCC (2006) methodology is referred to as a Tier 1 approach. The methodology described below shows how the Tier 1 and Tier 3 approaches can be combined to derive overall emission estimates. Refer to EPA (2010, 2007) for a complete description of the methodologies used to estimate N_2O emissions.

3.3.2.1 DAYCENT Simulations for Major Crop Types

The DAYCENT ecosystem model (Del Grosso et al. 2001, Parton et al. 1998) was used to estimate direct N_2O emissions from mineral soils producing major crops, (corn, soybean, wheat, alfalfa hay, other hay, sorghum, and cotton) which represent approximately 86% of total cropland in the United States. DAYCENT simulated crop growth, soil organic matter decomposition, greenhouse gas fluxes, and key biogeochemical processes affecting N_2O emissions. The simulations were driven by model input data generated from daily weather records, land management, and soil physical properties determined in national soil surveys.

DAYCENT simulations were conducted for each major crop at the county scale in the U.S. The county scale was selected because soil, weather, and crop area data were available for every county. However, land management data (e.g., timing of planting, harvesting, and fertilizer application; intensity of cultivation; rate of fertilizer application) were only available at the agricultural region level as defined by the Agricultural Sector Model (McCarl et al. 1993). There are 63 agricultural regions in the contiguous United States; most states correspond to one region, except for those with greater heterogeneity in agricultural practices, which led to further subdivisions. Therefore, while several cropping systems were simulated for each county in an agricultural region, the model parameters that determined the influence of management activities on soil N_2O emissions (e.g., when crops were planted/harvested, amount of fertilizer added), did not differ among those counties.

Corn, soybeans, wheat, alfalfa hay, other hay, sorghum, and cotton are defined as major crops and were simulated in every county where they were grown. For rotations that include a cycle that repeats every 2 or more years (e.g., corn/soybeans, wheat/corn/fallow) different simulations were performed where each phase of the rotation was simulated every year. For example, in regions where wheat/corn/fallow cropping is used, three rotations were simulated: one with wheat grown the first year, a second with corn the first year, and a third with fallow the first year. This ensured that each crop was represented during each year in one of the three simulations. In cases where the same crop was grown in the same year in two or more distinct rotations for a region, N_2O emissions were averaged across the different rotations to obtain a value for that crop. Emissions from cultivated fallow land were also included. Fallow area was assumed to be equal to winter wheat area in regions where winter wheat/fallow rotations are the dominant land management for winter wheat.

The simulations reported here assumed conventional tillage cultivation, gradual improvement of cultivars, and gradual increases in fertilizer application until 1989. We accounted for improvements of cultivars (cultivated varieties) because it is unrealistic to assume that modern corn is identical, in terms of yield potential, nitrogen demand, etc., as corn grown in 1900. Realistic simulations of historical land management and vegetation type are important because they influence present day soil carbon and nitrogen levels, which influence present day nitrogen cycling and associated N_2O emissions.

Nitrous oxide emission estimates from DAYCENT include the influence of N additions, crop type, irrigation, and other factors in aggregate, and therefore it is not possible to reliably partition N_2O emissions by anthropogenic activity (e.g., N_2O emissions from synthetic fertilizer applications cannot be distinguished from those resulting from manure applications). Consequently, emissions are not subdivided according to activity (e.g., N fertilization, manure amendments), as is suggested in the IPCC *Guidelines*, but the overall estimates are likely more accurate than the more simplistic IPCC method, which is not capable of addressing the broader set of driving variables influencing N_2O emissions. Thus, DAYCENT forms the basis for a more complete estimation of N_2O emissions than is possible with the IPCC methodology.

Uncertainty in the three major model inputs (weather, soil class, and N addition) was addressed using Monte Carlo analysis (Del Grosso et al. 2010). For example, although mean amounts of N fertilizer applied to different crops are known, the amounts of fertilizer applied by particular farmers are uncertain. Monte Carlo analysis provides a method to quantify how this type of uncertainty impacts N_2O emissions. There are three main steps in this analysis. First, a set of simulations was performed using mean N fertilizer additions, median weather, and the dominant soil texture class. These were designated the 0^{th} simulations. Second, probability distribution functions were derived for N additions, weather, and soil texture class. Third, Monte Carlo simulations were performed for a subset of counties in each agricultural region.

In addition to uncertainty in model inputs, model structural error was also addressed. Model structural error stems from models not being perfect representations of reality. That is, models contain assumptions and imperfectly represent the processes that control crop growth and N_2O emissions. To quantify model structural error, N_2O emissions generated by DAYCENT were compared with emissions measured in field plots at various locations in North America.

3.3.2.2 0th Simulations

For each crop in each county, simulations were performed assuming the most common land management practice, the weather most representative of the land area in the county where each crop is grown, and the most common soil type for the land area where each crop is grown (0th simulations). Simulations included native vegetation (year one to plow out), historical agricultural practices (plow out to 1970) and modern agriculture (1971 through 2008). Plow out (the year when native soils were initially cropped) was assumed to occur between 1600 and 1850, depending on the state in which the county lies. Simulation of at least 1,600 years of native vegetation was needed to initialize soil organic matter (SOM) pools in the model. Modern weather was used to drive the simulations of native vegetation and historical cropping. Simulation of native vegetation and the historical cropping period was needed to establish modern day SOM levels, which is important because N_2O emissions are sensitive to the amount of SOM. Annual model outputs for N_2O emissions, NO_3 leached/runoff, and N volatilized were compiled for the years 1990-2008.

3.3.2.3 Probability Distribution Functions

Probability distribution functions (PDFs) were derived for key model inputs, including weather, soil type, and N amendments. In each county selected for the Monte Carlo analysis, all of the 1 km^2 cells with daily weather that correspond to the land area where row crops and small grains dominate were identified and assigned an equal probability of being selected in an individual Monte Carlo simulation. Cells with daily weather were similarly identified for the areas cropped with hay. The three dominant soil map units were identified for the land area with row crops and small grains, and each was assigned a probability given their relative level of dominance. Three soil map units were similarly identified and assigned probabilities for the areas where hay dominates.

Mineral N fertilization rates were based on two sets of PDFs, which were specified for individual crop types and hay. The first PDF was the probability of a fertilization event and the second PDF was a log-normal distribution of fertilization rates. Both PDFs were derived from USDA surveys and supplemental information (ERS 1997, USDA NASS 2009, 2004, 1999, Grant & Krenz 1985). Irrigated and rain-fed crops were treated separately due to significantly different fertilization rates. State-level PDFs were developed for crops and hay if a minimum of 15 survey data points existed in the state. Where data were insufficient at the state-level, PDFs were developed for multi-state Farm Production Regions.

Uncertainty in manure amendments for crops and hay was incorporated in the analysis based on total manure available for application in each county, a weighted average amendment rate, and the crop-specific land area amended with manure for 1997 (Edmonds et al. 2003). Edmonds et al.

(2003) provided county-level estimates of the proportion of specific crops and hay land amended with manure in 1997. EPA (2010) provided supplemental data on county-level variation in manure production across the time series from 1990 to 2008. We used the EPA data to scale the amended area in 1997 for each crop and hay under the assumption that more manure production would increase the area amended with manure, and vice versa. The estimated area was then divided by the respective total areas in the county for each crop and hay, yielding a probability of either including a manure amendment or not in the Monte Carlo analysis. If soils were amended with manure, a reduction factor was applied to the N fertilization rate accounting for the interaction between fertilization and manure N amendments (i.e., farmers usually reduce mineral fertilization rates if applying manure). Reduction factors were randomly selected from PDFs based on relationships between manure N application and fertilizer rates (ERS 1997).

3.3.2.4 Monte Carlo Simulations

In each agricultural region, two counties were randomly selected for Monte Carlo simulations. Additional counties were selected based on the variance in N_2O emissions across regions from previous simulations (Del Grosso et al. 2010) by using a Neyman allocation (Cochran 1977). Neyman's optimization apportions samples based on an estimated variance in soil N_2O emissions. Using this approach, greater variance leads to a higher sampling density within the respective region with the goal of optimally capturing variation across the croplands in the conterminous U.S. Regions with greater variance in N_2O emissions were assumed to have more variability in weather, soil characteristics, and agronomic practices, suggesting that more counties needed to be included in the Monte Carlo analysis. In total, 300 counties were selected for the Monte Carlo simulations. As with the 0^{th} simulations, simulations of pre-settlement native vegetation and historical cropping patterns were performed in each county using the median weather for the county in combination with the three most dominant soil types.

One hundred Monte Carlo simulations were performed for each crop and hay type in the 300 counties selected for the Monte Carlo analysis. Random draws were made to select a soil type and weather file for the simulation from their respective PDFs, and the appropriate historical simulation was identified based on the soil type. Random draws were made to determine if mineral N fertilizer would be applied, the rate, and if the crop would be amended with manure. If manure was added, synthetic fertilizer rates were reduced based on an additional draw from the PDF for the reduction factors. The DAYCENT simulation was executed following the PDF draws and the process was repeated for a total of 100 iterations.

3.3.2.5 Nitrous Oxide Emission Estimates

Nitrous oxide emissions from the 0^{th} simulation for each crop in each county in each agricultural region were adjusted by comparing the 0^{th} simulation emissions to the mean emissions from the Monte Carlo simulations for that agricultural region. DAYCENT emissions for each crop in units of g N_2O-N m^{-2} were multiplied by the county-level crop area based on NASS data. Lastly, emissions from all crops were summed to obtain county-level and national emissions from cropped soils.

3.3.2.6 Activity Data for DAYCENT Simulations

The activity data requirements for estimating N_2O emissions from major crop types include the following: daily weather, soil texture, native vegetation, crop rotation and land management information, N fertilizer rates and timing, manure amendment N rates and timing, and county-level crop areas. Unlike the IPCC approach, N inputs from crop residues are not considered activity data in the DAYCENT analysis because N availability from this source is internally generated by the model. That is, while the model accounts for the contribution of crop residues to the soil profile and subsequent N_2O emissions, this source of mineral soil N is not activity data in the sense that it is not a model input.

Daily Weather Data: Daily maximum/minimum temperature and precipitation were obtained from the DAYMET model, which generates daily surface precipitation, temperature, and other meteorological data at 1 km^2 resolution driven by weather station observations and an elevation model (Thornton et al. 2000, 1997, Thornton & Running, 1999). DAYMET weather data are available for the United States at 1 km^2 resolution for 1980 through 2003.

Soil Properties: Soil texture data required by DAYCENT were obtained from STATSGO (Soil Survey Staff, USDA Natural Resources Conservation Service, 2005) and were based on observations. Observed data for soil hydraulic properties needed for model inputs were not available, so they were calculated from STATSGO texture class and Saxton et al.'s (1986) hydraulic properties calculator.

Native Vegetation by County: Pre-agricultural land cover for each county was designated according to the potential native vegetation used in the VEMAP (1995) analysis, which was based on the Kuchler (1964) Potential Vegetation Map for the conterminous United States.

Crop Rotation and Land Management Information by Agricultural Region: Data for the 63 agricultural regions were obtained for specific timing and type of cultivation, timing of planting/harvest, and crop rotation schedules (Hurd 1930, 1929, Latta 1938, Iowa State College Staff Members 1946, Bogue 1963, Hurt 1994, USDA 2000a, USDA 2000c, CTIC 1998, Piper et al. 1924, Hardies & Hume 1927, Holmes 1902, 1929, Spillman 1902, 1905, 1907, 1908, Chilcott 1910, Smith 1911, Kezer ca. 1917, Hargreaves 1993, ERS 2002, Warren 1911, Langston et al. 1922, Russell et al. 1922, Elliot & Tapp 1928, Elliot 1933, Ellsworth 1929, Garey 1929, Hodges et al. 1930, Bonnen & Elliot 1931, Brenner et al. 2001, 2002, Smith et al. 2002).

Nitrogen Fertilizer Amendment Rates and Timing by Agricultural Region: Fertilizer application rates and timing of applications within each of the 63 agricultural regions were determined from regional, state, or sub-state estimates for different crops. Estimates were obtained primarily from the USDA Economic Research Service Cropping Practices Survey (ERS 1997) with additional data from other sources, including the National Agricultural Statistics Service (USDA NASS 2009, 2004, 1999). Prior to 1990, estimates for crop-specific regional fertilizer rates were based largely on extrapolation/interpolation of fertilizer rates from the years with available data. For crops in some agricultural regions, little or no data were available, and therefore a geographic regional mean was used to simulate N fertilization rates.

Managed Livestock Manure[2] *Nitrogen Amendment Rates and Timing by Agricultural Region:* Data on managed manure N amendments to soils were available for 1997 (Kellogg et al. 2000), and demonstrated that less than half of manure N produced on an annual basis was applied to soils. Crop-specific application rates of manure N for other years between 1990 and 2008 were obtained by multiplying the 1997 crop-specific rates by the ratio of managed manure N produced in that year to the managed manure N produced in 1997; the amount of land receiving manure (approximately 5 percent of total cropped land) was assumed to be constant during 1990 through 2008. Nitrogen available for application was estimated for managed systems based on the total amount of N produced in manure minus N losses and including the addition of N from bedding materials. Nitrogen losses include direct nitrous oxide emissions, volatilization of ammonia and NO_x, and runoff and leaching. The remaining manure N that was not applied to major crops and grassland was assumed to be applied to non-major crop types. Manure was applied during spring at the same time as synthetic N fertilizer. Prior to 1990, manure application rates and timing were based on various sources (Brooks 1901, Anonymous 1924, Fraps & Asbury 1931, Ross & Mehring 1938, Saltzer & Schollenberger 1938, Alexander & Smith 1990). As with mineral N fertilization, data for manure were incomplete, so regional averages were used to fill spatial gaps in data and interpolation/extrapolation was used to fill temporal gaps. Manure N application rates during 1990 through 2004 were based on Kellogg et al. (2000).

Crop Areas by Crop Type and by County: County-level total crop area data were downloaded from the USDA NASS Web site for the years 1990 through 2008 (USDA NASS 2009), and these data formed the basis to scale emissions from individual crop types across the entire county.

3.3.3 IPCC Methodology for Non-Major Crop Types

3.3.3.1 Mineral Soils

For mineral agricultural soils producing non-major crop types, the Tier 1 IPCC methodology was used to estimate direct N_2O emissions. Estimates of direct N_2O emissions from N applications to non-major crop types were based on the annual increase in mineral soil N from the following practices: (1) the application of synthetic commercial fertilizers, (2) the retention of crop residues, and (3) manure and non-manure organic fertilizers.

IPCC methodology for emissions from mineral soils is based on nitrogen inputs. Nitrogen inputs from synthetic and organic fertilizer and aboveground and belowground crop residues were added together. This sum was multiplied by the IPCC default emission factor (1.0%) to derive an estimate of cropland direct N_2O emissions from non-major crop types. Nitrate leached or runoff and N volatilized from non-major crop types are calculated by multiplying N fertilizer applied by the IPCC (2006) default factors (30% and 10%, respectively).

[2] For purposes of the Inventory, total livestock manure is divided into two general categories: (1) managed manure, and (2) unmanaged manure. Managed manure includes manure that is stored in manure management systems such as pits and lagoons, as well as manure applied to soils through daily spread operations. Unmanaged manure encompasses all manure deposited on soils by animals on pasture, range, and paddock.

Annual synthetic fertilizer nitrogen additions to non-major crop types are calculated by process of elimination. For each year, fertilizer applied to major crops and grazed lands (as simulated by DAYCENT – approximately 80% of the U.S. total fertilizer used on farms) was subtracted from total fertilizer used on farms in the United States. The difference, approximately 20% of total synthetic fertilizer N used on farms in the U.S., was assumed to be applied to non-major crop types. Non-major crop types include fruits, nuts, and vegetables, which is estimated at approximately 5% of total U.S. N fertilizer use (TFI 2000), and other annual crops not simulated by DAYCENT, barley, oats, tobacco, sugarcane, sugar beets, sunflower, millet, peanuts, etc., which account for approximately 15% of total U.S. fertilizer used on farms. Manure N applied to non-major crops was estimated in a similar manner; manure applied to major crops and grazed lands as simulated by DAYCENT was subtracted from total manure available for soil application. This difference was assumed to be applied to non-major crops. In addition to synthetic fertilizer and manure N, nitrogen in soils due to the cultivation of non-major N-fixing crops (e.g., edible legumes) was included in these estimates. Finally, crop residue nitrogen was derived from information on crop production yields, residue management (retained vs. burned or removed), mass ratios of aboveground residue to crop product, dry matter fractions, and nitrogen contents of the residues (IPCC 2006). The activity data for these practices were obtained from the following sources:

- Annual production statistics for crops whose residues are left on the field: USDA (2003, 2002, 2001, 2000a, 1998, 1994), Schueneman (2001, 1999a, 1999b, 1999c), Deren (2002), Schueneman and Deren (2002), Cantens (2004), Lee (2004, 2003).
- Crop residue N was derived by combining amounts of above- and below-ground biomass, which were determined based on crop production yield statistics (USDA 2006, 2005, 2003, 1998, 1994), dry matter fractions (IPCC 2006), linear equations to estimate above-ground biomass given dry matter crop yields (IPCC 2006), ratios of below-to-above-ground biomass (IPCC 2006), and N contents of the residues (IPCC 2006).

Annual Applications of Commercial Non-Manure Organic Fertilizers by Agricultural Region: Estimates of total national annual N additions from land application of other organic fertilizers were derived from organic fertilizer statistics (TVA 1994, 1993, 1992a, AAPFCO 2006, 2005, 2004, 2003, 2002, 2000a, 2000b, 1999, 1998, 1997, 1996, 1995). The organic fertilizer data, which are recorded in mass units of fertilizer, had to be converted to mass units of N by multiplying by the average organic fertilizer N contents provided in the annual fertilizer publications. These N contents are weighted average values, and vary from year to year (ranging from 2.3 percent to 3.9 percent over the period 1990 through 2004). Annual on-farm use of these organic fertilizers is very small, less than 0.03 Tg N.

3.3.3.2 Cultivation of Histosols

The IPCC Tier 1 method is used to estimate direct N_2O emissions from the drainage and cultivation of organic cropland soils. Estimates of the total U.S. acreage of drained organic soils cultivated annually for temperate and sub-tropical climate regions was obtained for 1982, 1992, and 1997 from the Natural Resources Inventory (USDA 2000b, as extracted by Eve 2001 and amended by Ogle 2002), using temperature and precipitation data from Daly et al. (1998, 1994). To estimate annual N_2O emissions from histosol cultivation, the temperate histosol area is multiplied by the IPCC default emission factor for temperate soils (8 kg N_2O-N/ha cultivated;

IPCC 2006), and the sub-tropical histosol area is multiplied by the average of the temperate and tropical IPCC default emission factors (12 kg N_2O-N/ha cultivated; IPCC 2006).

3.3.3.3 Total N_2O Emissions

Total direct emissions were obtained by summing DAYCENT-generated emissions from major crops on mineral soils, IPCC-generated estimates for non-major crops on mineral soils, and IPCC estimates of emissions from organic soils. Total indirect emissions from NO_3 leaching or runoff were obtained by adding DAYCENT estimates for major crops on mineral soils to IPCC (2006) estimates for non-major crops on mineral soils and multiplying by the default emission factor (0.75% of N leached/runoff). Total indirect emissions from nitrogen volatilization were obtained by adding DAYCENT estimates for major crops on mineral soils to IPCC (2006) estimates for non-major crops on mineral soils and multiplying by the default emission factor (1% of N volatilized). Indirect emissions from NO_3 leaching or runoff were added to those from nitrogen volatilization to get total indirect emissions. Total direct and indirect emissions were then summed to get total N_2O emissions from cropped soils.

3.3.4 Uncertainty in N_2O Emissions

Uncertainty was estimated differently for each of the following components of N_2O emissions from cropped soils: direct emissions from major crops calculated by DAYCENT due to model input uncertainty, direct emissions from major crops calculated by DAYCENT due to model structure uncertainty, direct emissions from minor crops not calculated by DAYCENT, and indirect emissions from all crops. For direct emissions calculated using DAYCENT, model input uncertainty was quantified using the Monte Carlo analysis described above in section 3.3.2 and in more detail by Del Grosso et al. (2010). Model structure uncertainty was quantified by comparing DAYCENT estimates of N_2O emissions with measured values (Del Grosso et al. 2010). Uncertainty for direct emissions from minor crops was estimated using simple error propagation (IPCC 2006). Uncertainty in indirect emissions for major crops combined uncertainty in DAYCENT estimates of nitrate leaching and N gas volatilization based in the Monte Carlo simulations with uncertainty in the IPCC Tier 1 emissions factors used to convert these N loss vectors to N_2O emissions. Uncertainty in indirect emissions for minor crops combined uncertainty in IPCC Tier 1 emissions factors for nitrate leaching and N gas volatilization with uncertainty in the IPCC Tier 1 emissions factors used to convert these N loss vectors to N_2O emissions. Error propagation was used to combine uncertainties in the various components by taking the square root of the sum of the squares of the standard deviations of the components (IPCC 2006). The 95% confidence interval in N_2O emissions was estimated to lie between 114 and 241 Tg CO_2 eq. (Table 3-1).

3.3.5 Changes Compared to the 2nd Edition of the USDA GHG Report

Although there were no major changes in methodologies compared to the previous edition (USDA 2008), a series of improvements were implemented. Instead of assuming that nitrate leaching can occur anywhere, a criterion was used to designate lands where nitrate is susceptible to be leached into waterways, as suggested by IPCC (2006). This is based on observations that in semi-arid and arid areas, nitrate can be leached below the rooting zone, but it does not enter waterways because water tables in dry areas are low or non-existent. Other changes include:

using state-level N data for on-farm use of fertilizers to estimate synthetic N fertilizer application on non-major crops, including uncertainty in DAYCENT outputs of N volatilization and N leaching/runoff in the calculation of uncertainty for indirect emissions; using a default uncertainty of ±50 percent for Tier 1 uncertainties that were not addressed in the previous inventory (e.g., crop yields and organic fertilizer amendments); assuming that manure N available for land application not accounted for by the DAYCENT simulations was applied to non-major crop types; revising DAYCENT parameterization for sorghum; correcting an error in the empirically based uncertainty estimator; improved estimates of manure additions to croplands; and using sugar-cane-specific information for calculating the residue/crop ratio, fraction of residue burned, dry matter fraction, burning efficiency, and combustion efficiency for this crop. The main results of these changes are lower N_2O emissions and wider confidence intervals. Lower N_2O emission estimates were primarily due to the new operational version of DAYCENT and the revised structural uncertainty associated with the model. Earlier versions of DAYCENT tended to overestimate emissions, and although these emissions were adjusted using the structural uncertainty estimator, there was considerable uncertainty in those adjustments and it is likely that high estimates were not sufficiently adjusted downwards. The new operational version of DAYCENT does not systematically overestimate N_2O emissions for the majority of crops so overall emissions are lower. Including residual error from the linear mixed-effect model as a component of the structural uncertainty and accounting for additional sources of uncertainty mentioned above that were previously neglected are responsible for the wider uncertainty intervals.

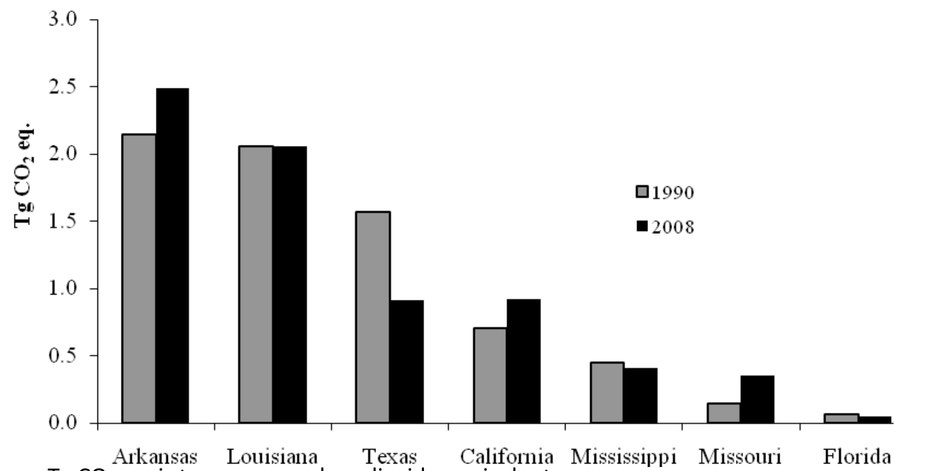
3.3.6 Mitigation of N_2O Emissions

Mitigation of N_2O emissions is based on optimizing the amount and timing of nitrogen fertilizer additions. Excess fertilizer applied to crops increases the nitrogen available for N_2O, N oxide and NH_3 emissions, and for NO_3 leaching. Using time-released fertilizers and applying fertilizer in multiple applications improves the synchrony between nitrogen supply and plant nitrogen demand. However, multiple applications of fertilizer require increased time and equipment usage by farmers and time-released fertilizers are more expensive than conventional fertilizers. Use of nitrification inhibitors has been shown to decrease N_2O emissions (Halvorson et al. 2010a, 2010b, Weiske et al. 2001, McTaggert et al. 1997). The capability to simulate their impact has been incorporated into the DAYCENT ecosystem model. National-scale DAYCENT simulations suggest that universal use of nitrification inhibitors could reduce total N_2O emissions by 10-20% while maintaining, or slightly increasing, crop yields. The model showed lower direct N_2O and NO_x emissions because nitrification rates were decreased, but also lower NO_3 leaching rates because reduced nitrification also reduces inputs to the soil NO_3 pool. However, fertilizer amended with nitrification inhibitors, as with time-released fertilizer, is more expensive. Further analyses of the environmental and economic costs and benefits of improved N source fertilizers need to be performed before optimum region-specific mitigation strategies can be identified.

3.4 Methane Emissions from Rice Cultivation

Methane emissions from rice cultivation[3] are limited to seven U.S. states (Figure 3-3). In four states (Arkansas, Florida, Louisiana, and Texas), the climate allows for cultivation of two rice crops per season, the second of which is referred to as a ratoon crop (EPA 2010). Methane emissions from primary and ratoon crops are accounted for separately because emissions from ratoon crops are higher (EPA 2010). Overall, rice cultivation is a small source of CH_4 in the United States. In 2008, CH_4 emissions totaled 7.2 Tg CO_2 eq, of which 5.3 Tg CO_2 eq. were from primary crops in all seven states and 1.9 Tg CO_2 was from ratoon crops in four states (Table 3-4).

Figure 3-3
Methane from Rice Cultivation by State, 1990 & 2008



Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Table 3-4 Methane from Rice Cultivation from Primary and Ratoon Operations by State, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Source						*Tg CO2 eq.*					
Primary	**5.1**	**5.6**	**5.5**	**5.9**	**5.7**	**5.4**	**6.0**	**6.0**	**5.1**	**4.9**	**5.3**
Arkansas	2.1	2.4	2.5	2.9	2.7	2.6	2.8	2.9	2.5	2.4	2.5
California	0.7	0.8	1.0	0.8	0.9	0.9	1.1	0.9	0.9	1.0	0.9
Florida	0.02	0.04	0.03	0.02	0.02	0.01	0.02	0.02	0.02	0.03	0.02
Louisiana	1.0	1.0	0.9	1.0	1.0	0.8	1.0	0.9	0.6	0.7	0.8
Mississippi	0.4	0.5	0.4	0.5	0.5	0.4	0.4	0.5	0.3	0.3	0.4
Missouri	0.1	0.2	0.3	0.4	0.3	0.3	0.3	0.4	0.4	0.3	0.4
Texas	0.6	0.6	0.4	0.4	0.4	0.3	0.4	0.4	0.3	0.3	0.3
Ratoon	**2.1**	**2.1**	**2.0**	**1.7**	**1.1**	**1.5**	**1.6**	**0.8**	**0.9**	**1.3**	**1.9**
Arkansas	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Florida	0.04	0.08	0.05	0.05	0.04	0.04	0.05	0.00	0.02	0.03	0.03
Louisiana	1.1	1.1	1.3	1.1	0.5	1.0	1.1	0.5	0.5	0.9	1.2
Texas	0.9	0.8	0.7	0.6	0.5	0.5	0.5	0.4	0.4	0.3	0.6
Total	**7.1**	**7.6**	**7.5**	**7.6**	**6.8**	**6.9**	**7.6**	**6.8**	**5.9**	**6.2**	**7.2**

Note: Tg CO_2 eq. is teragrams carbon dioxide equivalent.

[3] This source focuses on CH_4 emissions resulting from anaerobic decomposition and does not include emissions from burning of rice residues. The latter is covered in section 3.5.

Arkansas and Louisiana had the highest CH_4 emissions (2.91 Tg CO_2 eq. and 1.4 Tg CO_2 eq. respectively) from rice cultivation in 2008, followed by California and Texas. Mississippi, Missouri, and Florida each had emissions less than or equal to 0.5 Tg CO_2 eq. (Table 3-4). Overall since 1990, CH_4 emissions from rice cultivation have increased by 1% (Table 3-5). While national-scale changes were small between 1990 and 2008 (1% increase), sizeable shifts occurred at state levels during that time period. For example, CH_4 emissions in Missouri and California increased by 149% and 31%, respectively, while emissions in Texas declined by 42% (Table 3-5). Although CH_4 emissions from Missouri increased by 149% between 1990 and 2008, they remained small in magnitude relative to emissions from other states because of the small land area used for rice production in this state. State-level shifts in CH_4 emissions since 1990 are positively correlated with changes in area of rice cultivation (Appendix Table B-1). Appendix Table B-1 provides a complete time series of areas harvested for rice by state with primary versus ratoon crops from 1990-2008.

3.4.1 Methods for Estimating CH_4 Emissions from Rice Cultivation

Table 3-5 Change in Methane Emissions from Rice Cultivation, 1990-2008

	1990	2008	1990-2008
State	*Tg CO_2 eq.*		*% Change*
Arkansas	2.14	2.49	16%
California	0.70	0.92	31%
Florida	0.06	0.05	-19%
Louisiana	2.06	2.06	0%
Mississippi	0.45	0.41	-8%
Missouri	0.14	0.36	149%
Texas	1.57	0.91	-42%
Total	**7.12**	**7.20**	**1%**

Note: Tg CO_2 eq. is teragrams carbon dioxide equivalent.

The EPA provided estimates for CH_4 emissions from rice cultivation for this report. Details on the methods are provided below and are excerpted, with permission from EPA, from Chapter 6 of the U.S. GHG Inventory report (EPA 2010). The method used by EPA applies area-based seasonally integrated emission factors (i.e., amount of CH_4 emitted over a growing season per unit harvested area) to harvested rice areas to estimate annual CH_4 emissions from rice cultivation. The EPA derives specific CH_4 emission factors from published studies containing rice field measurements in the United States, with separate emissions factors for ratoon and primary crops to account for higher seasonal emissions in ratoon crops.

A review of published experiments was used to develop emissions factors for primary and ratoon crops. Experiments where nitrate or sulfate fertilizers or other substances believed to suppress CH_4 formation were applied, and experiments where measurements were not made over an entire flooding season or where floodwaters were drained mid-season, were excluded from the analysis. The remaining experimental results were then sorted by season (i.e., primary and ratoon) and type of fertilizer amendment (i.e., no fertilizer added, organic fertilizer added, and synthetic and organic fertilizer added). The experimental results from primary crops with synthetic and organic fertilizer added (Bossio et al. 1999, Cicerone et al. 1992, Sass et al. 1991a and 1991b) were averaged to derive an emission factor for the primary crop, and the experimental results from ratoon crops with synthetic fertilizer added (Lindau et al. 1995, Lindau & Bollich 1993) were averaged to derive an emission factor for the ratoon crop. The resultant emission factor for the primary crop is 210 kg CH_4/ha per season, and the resultant emission factor for the ratoon crop is 780 kg CH_4/ha per season.

The harvested rice areas for the primary and ratoon crops in each state are presented in Appendix Table B-1. Primary crop areas for 1990 through 2008 for all states except Florida and Oklahoma were taken from USDA NASS Field Crops Final Estimates 1987-1992 (USDA 1994), Field Crops Final Estimates 1992-1997 (USDA 1998a), Crop Production 2000 Summary (USDA 2003), and Crop Production 2001 Summary (USDA 2005-2009). Harvested rice areas in Florida, which are not reported by USDA, were obtained from Tom Schueneman (2001, 2000, 1999b, 1999c), a Florida agricultural extension agent, and Chris Deren (2002) of the Everglades Research and Education Center at the University of Florida. Acreages for the ratoon crops were derived from conversations with the agricultural extension agents in each state. California, Mississippi, Missouri, and Oklahoma have not ratooned rice over the period 1990 through 2008 (Guethle 1999, 2000, 2001, 2002 through 2008; Lee 2003 through 2007; Mutters 2002 through 2005; Street 1999 through 2003; Walker 2005, 2007, 2008). In Arkansas, ratooning occurred only in 1998 and 1999, when the ratoon area was less than 1% of the primary area (Slaton 2001, 2000, 1999). In Florida, the ratoon area was 50% of the primary area from 1990 to 1998 (Schueneman 1999a), about 65% of the primary area in 1999 (Schueneman 2000), around 41% of the primary area in 2000 (Schueneman 2001), and about 70% of the primary area in 2001(Deren 2002). In Louisiana, the percentage of the primary area in ratoon was constant at 30% over the 1990 to 1999 period, but increased to approximately 40% in 2000 before returning to 30% in 2001 (Linscombe 2002, 2001,1999a, Bollich 2000). In Texas, the percentage of the primary area in ratoon was constant at 40% over the entire 1990 to 1999 period and in 2001, but increased to 50% in 2000 due to an early primary crop (Klosterboer 2002, 2001, 2000, 1999a,1999b).

3.4.2 Uncertainty in Estimating Methane Emissions from Rice Cultivation

The following discussion of uncertainty in estimating GHG emissions from rice cultivation is modified from information provided in the U.S. GHG Inventory (EPA 2010). The information is reproduced here with permissions from the EPA.

Methane emission factors are the largest source of uncertainty in estimates for rice cultivation. Seasonal emissions, derived from field measurements in the United States, vary by more than an order of magnitude resulting from a variation in cultivation practices, fertilizer applications, cultivar types, soil, and climatic conditions. Some variability is accounted for by separating primary from ratoon areas. However, even within a cropping season, measured emissions vary significantly. Of the experiments that were used to derive the emission factors used here, primary emissions ranged from 22 to 479 kg CH_4/ha per season and ratoon emissions ranged from 481 to 1,490 kg CH_4/ha per season.

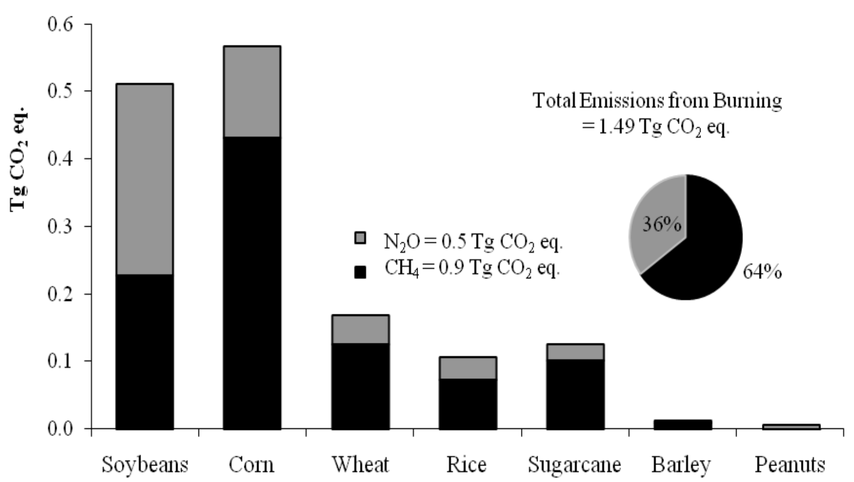
Data are not collected regularly on the area of rice crops in ratoon, creating another source of uncertainty. The area estimates are derived from expert opinion and account for 1 to 5% of the total area of rice cultivation. A final source of uncertainty is the practice of flooding outside of the normal rice season. According to agriculture extension agents, this occurs in all rice-growing states. No uncertainties were calculated for the practice of flooding outside of the normal rice season because CH_4 flux measurements have not been undertaken over a sufficient geographic range or under a broad enough range of representative conditions to account for this source in the emission estimates or its associated uncertainty.

To quantify the uncertainties for emissions from rice cultivation, a Monte Carlo (Tier 2) uncertainty analysis was performed using the information provided above. The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 3-1. Rice cultivation CH4 emissions in 2008 were estimated to be between 2.6 and 17.5 Tg CO_2 eq. at a 95-percent confidence level, which indicates a range of 64 percent below to 143 percent above the actual 2008 emission estimate of 7.2 Tg CO_2 eq.

3.5 Residue Burning

Greenhouse gas emissions from field burning of crop residues are a function of the amount and type of residues burned. In the U.S., crops burned include wheat, rice, sugarcane, corn, barley, soybeans, and peanuts (EPA 2010). For most crops, less than 5% of residues are burned per year, but a higher portion of rice residues is burned annually (EPA 2010). Consequently, emissions from residue burning are a small source of overall crop-related emissions in the U.S. About three-fifths of GHG emissions from residue burning, across all crop types, consisted of CH_4 in 2008; the remaining was N_2O (Table 3-6, Figure 3-4). CO_2 burning is not considered a GHG source because the CO_2 lost from burning was assimilated during the year by growing vegetation. The highest GHG emissions were from burning of corn and soybean crop residues, at 38 and 34% respectively. Burning of wheat, rice, sugarcane, and barely crop residues each contributed 11% or less to overall GHG emissions; burning of peanut crop residues contributed almost nothing to this source of GHG due to the relatively small amount of land area planted with this crop.

Figure 3-4
Greenhouse Gas Emissions from Field Burning by Crop Type, 2008



Tg CO2 eq. is teragrams carbon dioxide equivalent; N2O is nitrous oxide; CH4 is methane.

Total greenhouse gas emissions from residue burning increased 29% from 1990 to 2008. Trends in relative GHG emissions were similar across crop types in 1990 compared to 2008 with a few exceptions. In both 1990 and 2008, burning of corn residues contributed the most to GHG emissions from residue burning, while burning of soybeans was the second largest source. Between 1990 and 2008, soybean and corn production both increased in absolute amounts (Figure 3-5). However, proportionally, soybean production increased slightly more than corn (soybean production increased by 54% and corn by 53%) (Figure 3-6). Despite the higher nitrogen content in soybeans relative to corn, corn production was still greater than soybean production in 2008, thus resulting in higher GHG emissions from residue burning.

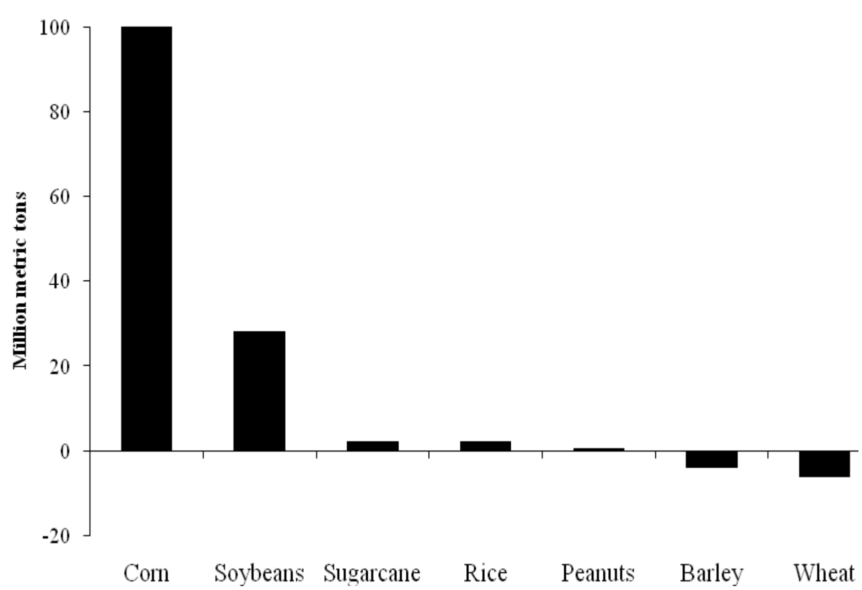
Table 3-6 Greenhouse Gas Emissions from Agriculture Burning by Crop, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Source						*Tg CO2 eq.*					
CH_4	**0.77**	**0.75**	**0.89**	**0.87**	**0.80**	**0.89**	**0.96**	**0.93**	**0.90**	**0.96**	**0.97**
Wheat	0.14	0.11	0.11	0.10	0.08	0.12	0.11	0.11	0.09	0.10	0.13
Rice	0.09	0.09	0.08	0.08	0.06	0.10	0.08	0.09	0.09	0.08	0.07
Sugarcane	0.09	0.10	0.12	0.11	0.12	0.11	0.10	0.09	0.10	0.10	0.10
Corn	0.28	0.26	0.35	0.34	0.32	0.36	0.42	0.40	0.37	0.46	0.43
Barley	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Soybeans	0.15	0.17	0.21	0.22	0.21	0.19	0.24	0.24	0.25	0.21	0.23
Peanuts	0.002	0.002	0.002	0.003	0.002	0.003	0.003	0.003	0.002	0.002	0.003
N_2O	**0.39**	**0.40**	**0.48**	**0.49**	**0.45**	**0.47**	**0.53**	**0.52**	**0.52**	**0.50**	**0.52**
Wheat	0.05	0.04	0.04	0.03	0.03	0.04	0.04	0.04	0.03	0.04	0.04
Rice	0.04	0.04	0.04	0.04	0.03	0.05	0.04	0.04	0.04	0.04	0.03
Sugarcane	0.021	0.023	0.028	0.026	0.027	0.026	0.022	0.020	0.023	0.023	0.023
Corn	0.09	0.08	0.11	0.11	0.10	0.11	0.13	0.12	0.12	0.15	0.14
Barley	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Soybeans	0.18	0.21	0.26	0.28	0.26	0.23	0.30	0.29	0.31	0.26	0.28
Peanuts	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.002	0.001	0.001	0.002
Total	**1.16**	**1.14**	**1.37**	**1.35**	**1.26**	**1.36**	**1.49**	**1.45**	**1.43**	**1.46**	**1.49**

Note: Tg CO_2 eq. is teragrams carbon dioxide equivalent; CH_4 is methane; N_2O is nitrous oxide.

Appendix Table B-2 provides the complete time series of crop production from 1990 to 2008 for crop types that contribute to GHG emissions from burning. Appendix Table B-3 provides nationwide data for crop production managed with burning by year. Production of crops such as corn and soybeans has been slowly increasing since 1990, with other crops like wheat, rice, and sugarcane remaining relatively constant or decreasing. Barley production has declined since the mid-1990s.

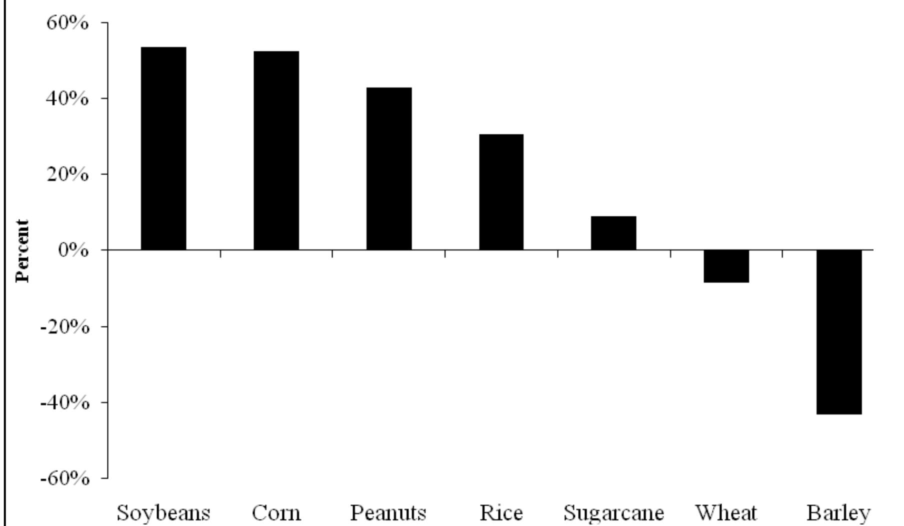
Figure 3-5
Change in Commodity Production, 1990-2008



The state-level rice harvest estimates were provided directly by EPA based on state production data.

3.5.1 Methods for Estimating CH_4 and N_2O Emissions from Residue Burning

Figure 3-6
Change in Commodity Production, 1990-2008



EPA provided national-level estimates of GHG emissions from agricultural residue burning for all crop types, and state-level estimates for GHG emissions from rice residue burning for this report. In addition, state-level estimates were derived by USDA for all crop types (except rice) using the same method. Details on the methods used by EPA are provided below, including excerpts from Chapter 6 of the U.S. GHG Inventory report (EPA 2010). This information is reproduced with permission from EPA.

The equations below were used to estimate the amounts of carbon and nitrogen released during burning.

Carbon Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned in situ) × (Dry Matter Content of the Residue) × (Burning Efficiency) × (Carbon Content of the Residue) × (Combustion Efficiency)

Nitrogen Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned in situ) × (Dry Matter Content of the Residue) × (Burning Efficiency) × (Nitrogen Content of the Residue) × (Combustion Efficiency)

Values used in the above equations to estimate emissions from residue burning are summarized in Appendix Table B-4. National and state-level crop production statistics are provided in Appendix Table B-2 and Appendix Table B-3. The sources for developing these input data are described for each parameter below.

Annual Crop Production:
Crop production data for all crops except rice in Florida and Oklahoma were taken from the USDA's Field Crops, Final Estimates 1987–1992, 1992–1997, 1997–2002 (USDA 1994, 1998, 2003), and *Crop Production Summary* (USDA 2005 through 2009). Rice production data for Florida and Oklahoma, which are not collected by USDA, were estimated separately. Average primary and ratoon crop yields for Florida (Schueneman & Deren 2002) were applied to Florida

acreages (Schueneman 1999b, 2001; Deren 2002; Kirstein 2003, 2004; Cantens 2004, 2005; Gonzalez 2007a, 2008, 2009), and crop yields for Arkansas (USDA 1994, 1998, 2003, 2005 through 2009) were applied to Oklahoma acreages (Lee 2003 through 2006; Anderson 2008, 2009).

Residue-to-Crop Product Mass Ratios:
All residue/crop product mass ratios except sugarcane were obtained from Strehler and Stützle (1987) The ratio for sugarcane is from Kinoshita (1988).

Fraction of Residues Burned:
The percentage of crop residue burned was assumed to be 3 percent for all crops in all years, except rice and sugarcane, based on state inventory data (Noller 1996, Cibrowski 1996, Oregon Department of Energy 1995, ILENR 1993, and the Wisconsin Department of Natural Resources 1993). Estimates of the percentage of rice residue burned were derived from state-level estimates of the percentage of rice area burned each year, which were multiplied by state-level annual rice production statistics. The annual percentages of rice area burned in each state were obtained from agricultural extension agents in each state and reports of the California Air Resources Board (Buehring 2009; Guethle 2009, 2008, 2007; Lancero 2006 through 2009; Texas Agricultural Experiment Station 2006 through 2009; Wilson 2003, 2004, 2005, 2007, and 2009; Lee 2005 through 2007; Sacramento Valley Basinwide Air Pollution Control Council 2005 and 2007; Walker 2004 through 2008; anonymous 2006; Cantens 2005; Stansel 2004, 2005; Lindberg 2002, 2003; Deren 2002; Najita 2001 and 2000; California Air Resources Board 2001, 1999; Bollich 2000; Fife 1999; Street 2001 through 2003; Klosterboer 2000 through 2003, 1999a, 1999b; Linscombe 2001 through 2009, 1999a, 1999b; Schueneman 2001, 1999a, 1999b). The estimates provided for Florida remained constant over the entire 1990 through 2008 period. While the estimates for all other states varied over the time series, estimates for Missouri remained constant through 2005, dropped in 2006, and remained constant near the 2006 value in 2007 and 2008. For California, the annual percentages of rice area burned in the Sacramento Valley are assumed to be representative of burning in the entire state, because the Sacramento Valley accounts for over 95 percent of the rice acreage in California (Fife 1999). These values generally declined between 1990 and 2008 because of a legislated reduction in rice straw burning (Lindberg 2002), although there was a slight increase from 2004 to 2005 and from 2006 to 2007. Estimates for percent of sugarcane burned were obtained from Ashman (2008).

Residue Dry-Matter Content:
Residue dry-matter contents for all crops except soybeans and peanuts were obtained from Turn et al. (1997). Soybean dry-matter content was obtained from Strehler and Stützle (1987). Peanut dry-matter content was obtained through personal communications with Jen Ketzis (1999), who accessed Cornell University's Department of Animal Science's computer model, Cornell Net Carbohydrate and Protein System.

Burning and Combustion Efficiency:
Burning efficiency refers to the fraction of dry biomass exposed to burning that actually burns and the combustion efficiency refers to the fraction of carbon in the fire that is oxidized completely to CO_2. The burning efficiency was assumed to be 93% and the combustion efficiency was assumed to be 88%, for all crop types, except sugarcane (EPA 1994). For

sugarcane, the burning efficiency was assumed to be 81% (Kinoshita 1988) and the combustion efficiency was assumed to be 68% (Turn et al. 1997). Emission ratios and conversion factors for all gases were taken from IPCC Guidelines (1996).

Carbon and Nitrogen Content:
The residue carbon contents and nitrogen contents for all crops except soybeans and peanuts are from Turn et al. (1997). The residue carbon content for soybeans and peanuts is the IPCC default (IPCC UNEP OECD IEA 1997). The nitrogen content of soybeans is from Barnard and Kristoferson (1985) and the nitrogen content of peanuts is from Ketzis (1999).

3.5.2 Uncertainty in Estimating Methane and Nitrous Oxide Emissions from Residue Burning

The following discussion of uncertainty in estimating GHG emissions from residue burning is modified from information provided in the U.S. GHG Inventory (EPA 2010). The information is reproduced here with permission from EPA.

Assumptions about the annual amount of residues burned by crop type are the largest source of uncertainty in estimating GHG emissions from field burning of agricultural residues. Data on the fraction burned, as well as the gross amount of residue burned each year, is not collected at either the national or state level. In addition, burning practices are highly variable among crops and states. The fractions of residue burned used in these calculations are based upon information collected by state agencies and in published literature. These emissions estimates may continue to change as more information becomes available in the future. Other sources of uncertainty include the residue/crop product mass ratios, residue dry matter contents, burning and combustion efficiencies, and emission ratios. Residue/crop product ratios for specific crops can vary among cultivars and, for all crops except sugarcane, generic global residue/crop product ratios were used rather than ratios specific to the United States. In addition, residue dry matter contents, burning and combustion efficiencies, and emission ratios can vary due to weather and other combustion conditions, such as fuel geometry. Values for these variables were taken from literature on agricultural biomass burning.

A Monte Carlo analysis was performed to quantify the uncertainties mentioned above. The calculated 95% confidence interval was 0.2 to 1.0 Tg CO^2 eq. for N_2O emissions from residue burning, or 71% below and 83% above the estimate of 0.5 Tg CO^2 eq. and 0.3 to 1.8 Tg CO^2 eq. for CH_4 emissions from residue burning, or 68% below and 88% above the estimate of 1.0 Tg CO^2 eq. (Table 3-1).

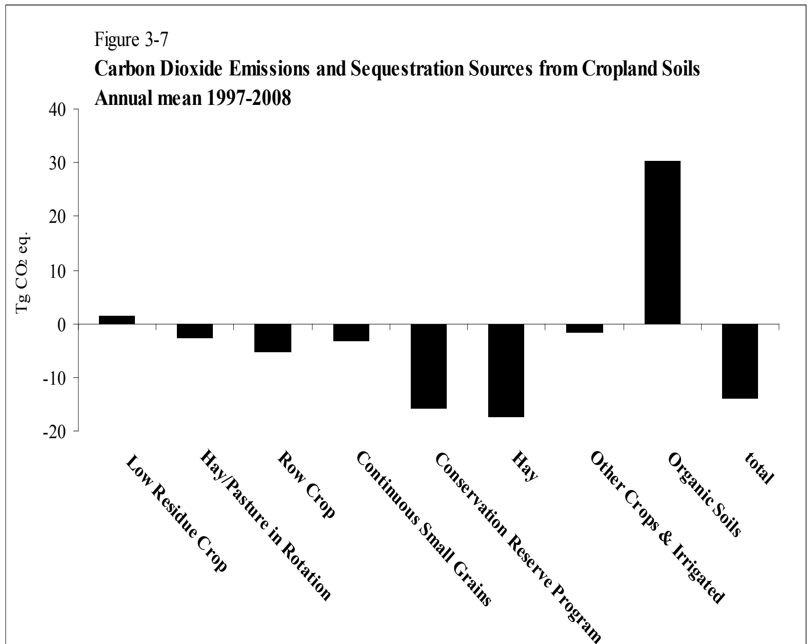
3.6 Carbon Stock Changes in Cropped Soils

Except for cultivated organic soils and liming practices, cropped soils in the U.S. were estimated to accumulate about 42 Tg CO_2 eq. in 2008 (Table 3-1)[4]. Much of the carbon change is

[4] Emissions and sinks of carbon in agricultural soils are expressed in terms of CO_2 equivalents; carbon sequestration is a result of changes in stocks of carbon in soils, from which CO_2 fluxes are inferred. Units of CO_2 equivalent can be converted to carbon using a multiplier of 0.272.

attributable to the Conservation Reserve Program, land use conversions between annual croplands and perennial hay and grazing lands, and land management (Figure 3-7). Practices such as the adoption of conservation tillage, including no-till, which have taken place over the past two decades, and reduced frequency of summer-fallow are important drivers of carbon stock changes. Manure applications to cropland also impact the estimated soil carbon stock increase.

In contrast, the small area of cultivated organic soils – less than 1 million hectares of a total 386 million hectares of agricultural and forest land – concentrated in Florida, California, the Gulf and Southeastern coastal region, and parts of the upper Midwest was a net source of CO_2 emissions for all years covered by the inventory (1990-2008). In 2008, about 30 Tg CO_2 eq. was emitted from cultivation of these soils (Table 3-1). Liming of agricultural soils resulted in emissions of about 4 Tg CO_2 eq per year. Total net carbon sequestration in 2008 equaled ~8 Tg CO_2 eq. when all of the above components were taken into consideration. Carbon uptake on agricultural soils varied between 1990 and 2008 (Table 3-2), driven largely by land use changes and weather fluctuations.



Tg CO2 eq. is teragrams carbon dioxide equivalent.

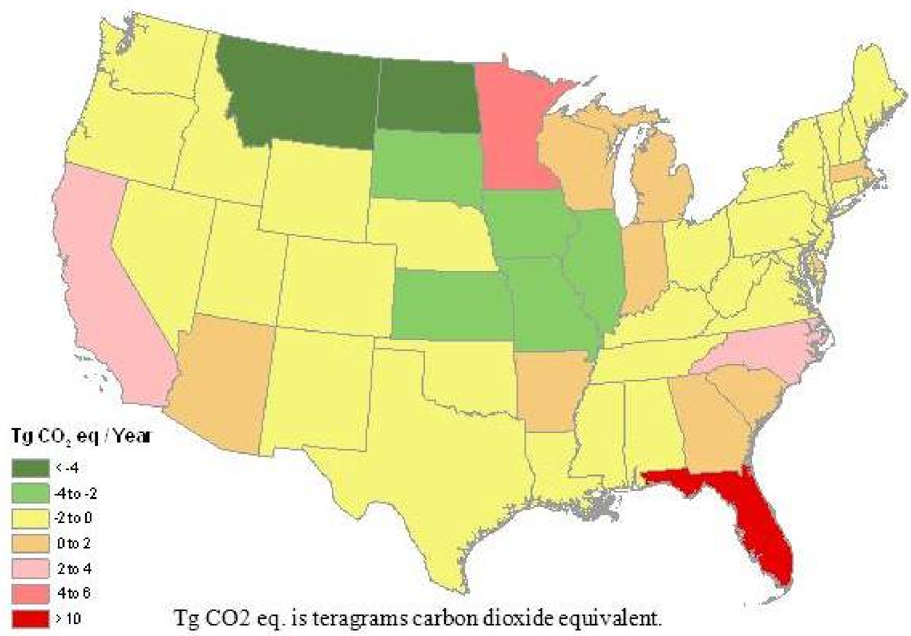
Most states in the Corn Belt and northern Great Plains are storing C in cropped soils due to adoption of reduced tillage and other practices (Map 3-3). The exception to this is Minnesota, which is losing C at the state level. Carbon losses from cropping of organic soils exceed C gains in mineral soil cropping for this state. Florida has the highest C losses, primarily due to sugarcane cropping on organic soils.

3.6.1 Methods for Estimating Carbon Stock Changes in Agricultural Soils

Two broad categories of cropland were considered, cropland remaining cropland and land converted to cropland. Within both of these categories, Tier 2 and Tier 3 methodologies were used. The Tier 2 approach is based on relatively simple equations used in IPCC (2003) methodology that have been modified to better represent nations or regions within nations. The Tier 3 approach (CENTURY model) uses a more complex ecosystem model to simulate carbon fluxes for cropped systems. Both tiers used land use and management data based primarily on the National Resources Inventory (NRI) (USDA 2000b). The NRI represents a robust statistical sampling of land use and management on all non-federal land in the United States, and greater than 400,000 NRI survey points occurred in agricultural lands and were used in the inventory analysis. The methodology summarized below is described in detail in the U.S. GHG Inventory (EPA 2010).

3.6.2.1 CENTURY Model Simulations for Most Cropped Mineral Soils

Map 3-3
State-Level Carbon Dioxide Fluxes from Cropped Soils in 2008.



CENTURY simulates carbon and nitrogen dynamics, soil water content and temperature, and other ecosystem variables (Parton et al.1994). Key submodels include: plant growth, senescence of biomass, decomposition of dead plant material and soil organic matter, and mineralization of nitrogen. Model inputs are monthly maximum/minimum air temperature and precipitation, surface soil texture class, soil hydric condition, vegetation type, and land management information (e.g., cultivation timing and intensity, timing and amount of fertilizer and organic matter amendments). Soil organic matter is simulated to a depth of 20 cm, while water, temperature, and mineral nitrogen are simulated throughout the soil profile. Soil organic matter is divided into three pools based on decomposability: active (turns over in months to years), slow (turns over in decades), and passive (turns over in centuries). The model accounts for the effects of nutrient availability, water, and temperature on plant growth (CO_2 uptake) and the effects of these factors, as well as cultivation, on decomposition (CO_2 release). The ability of the model to integrate carbon gains and losses and simulate plant growth and soil carbon levels reliably has been demonstrated using data from many sites in the U.S. and around the world (Parton et al.1994, Cerri et al. 2007, Ogle et al. 2007). The model has been shown to work in all the major biomes of the earth and can accurately reproduce the impacts of climate, soil texture, and land management on carbon fluxes (Parton et al. 1993, Kelly et al. 1997, Lugato 2007, Bricklemyer 2007). CENTURY has been parameterized to represent the major crops grown in the U.S. The major crops simulated by CENTURY for this analysis were corn, soybeans, small grains, hay, sorghum, millet, and cotton, which cover ~90% of U.S. cropland. Crops not simulated by CENTURY include; rice, sugarcane, tobacco, vegetables, orchards, and horticultural crops.

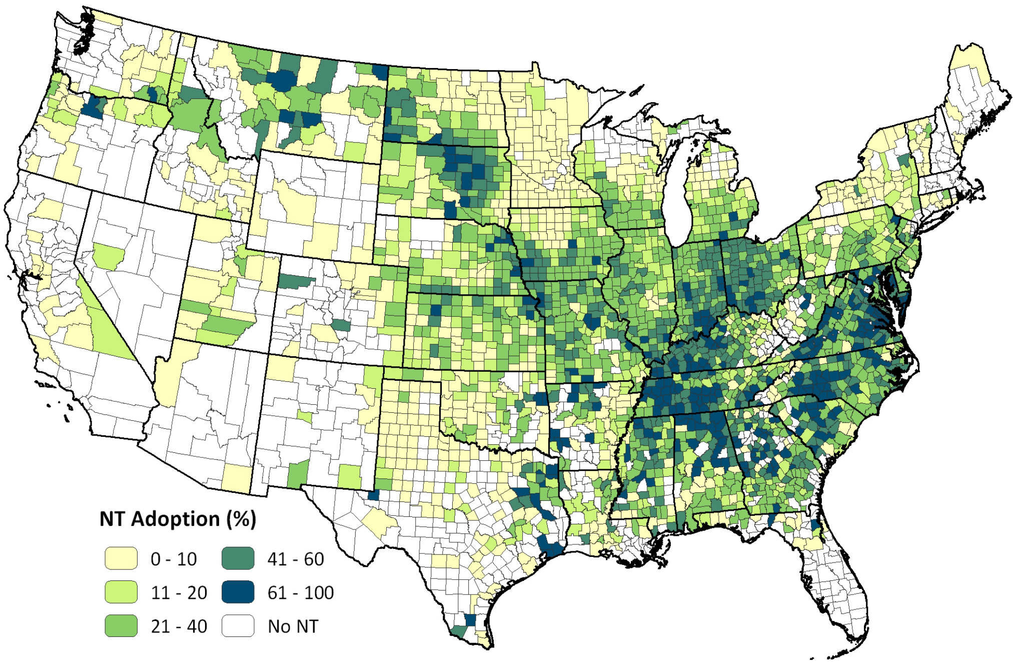
Three sets of simulations were performed: one to represent pre-settlement native vegetation, one to represent historical cropping, and one to represent modern cropping. This is important because previous vegetation types and land management activities influence the capacity of present day soils to lose or sequester carbon. Native vegetation was represented at the MLRA (Major Land Resource Area, USDA NRCS 1981) level. MLRAs represent geographical units with relatively similar soils, climate, water resources, and land use. Data on historical cropping practices for different regions were obtained from various sources including historical accounts and from NASS. Beginning in 1979, the first year of the NRI survey, simulations of crops and management practices were based on NRI data. Additional data for tillage practices used (Maps 3-4, 3-5) were from the Conservation Technology Information Center (CTIC 1998). Crop-specific N fertilization rates were from the USDA Economic Research Service survey (ERS 1997) and other sources (e.g., NASS). Manure application rates were estimated from data compiled by the USDA Natural Resources Conservation Service (Edmonds et al. 2003). Monthly weather data required to run CENTURY were from the PRISM database. PRISM (Daly et al. 1994) is based on observed weather, and the resolution is 4x4 km grid cells. The data were area weighted to represent the agricultural land in each county in the U.S. Soil texture and drainage capacity (hydric vs. non-hydric) were derived from the NRI.

3.6.2.2 Tier 2 Approach for Remaining Cropped Mineral Soils, Organic Soils, and Liming

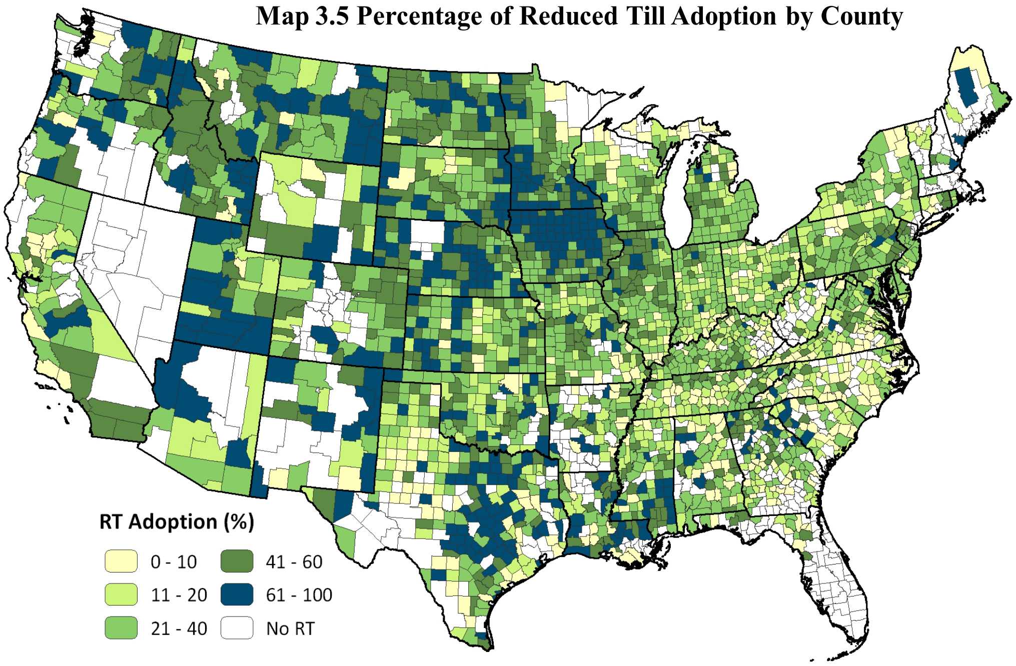
A Tier 2 approach was used to estimate soil carbon stock changes for crops not simulated by the CENTURY model, for non-agricultural lands that were converted to cropland, and for organic soils. Data on climate, soil type, and land use were used to classify land area and to apply appropriate stock change factors. U.S.-specific carbon stock change factors were derived from published literature to estimate the impact of management practices (e.g., changes in tillage or crop rotation) on soil carbon fluxes (Ogle et al. 2006b, 2003). Cultivated histosol areas are listed in Appendix Table B-5, carbon loss rates from organic soils under agricultural management in the United States are listed in Appendix Table B-6, state-level estimates of annual soil carbon stock changes by major land use and management type are listed in Appendix Table B-7, and state-level estimates of mineral soil carbon changes on cropland by major activity are listed in Appendix Table B-8.

Stock change factors and reference carbon stocks can vary for different climate regimes and soil types. The IPCC method defines eight climate types according to mean annual temperature, precipitation, and potential evapotranspiration. Six of these occur in the continental United States. The PRISM long-term monthly climate data set (Daly et al. 1998) was used to classify each of the 180 Major Land Resource Areas (MLRAs) in the United States into climate zones.

Map 3.4 Percentage of No-Till Adoption by County



Map 3.5 Percentage of Reduced Till Adoption by County



Reference soil carbon stocks were stratified by climate region and categorized into six major groupings, based on taxonomic orders that relate to soil development and physical characteristics that influence soil carbon contents. Estimates for carbon stocks under conventionally managed cropland (defined as the reference land use) were derived from the National Soil Survey Characterization Database (USDA NRCS 1997).

Based on the NRI, crop management systems were aggregated into 22 different categories. State-level estimates of mineral soil carbon changes on cropland by major activity are listed in Appendix Table B-9. Tillage practices are not included in the NRI. Thus, supplemental data were used from the Conservation Technology Information Center (CTIC 1998), which provides spatial information on tillage practices (Maps 3-4, 3-5). Data for wetland restoration under the Conservation Reserve Program (CRP) were obtained from Euliss and Gleason (2002). Manure N amendments over the inventory time period were based on application rates and areas amended with manure N from Edmonds et al. (2003).

Organic soils (i.e., peat, mucks) that have been drained and converted to cropland or pasture are subject to potentially high rates of carbon loss. Annual C losses were estimated using IPCC (2006, 1997) methodology, except that U.S.-specific carbon loss rates were used in the calculations instead of the default IPCC rates (Ogle et al. 2003).

Limestone and dolomite are often applied to acidic soils to raise the pH. However, CO_2 is emitted when these materials degrade. Emissions were estimated using a Tier 2 approach. Application rates were derived from estimates and industry sources (Minerals Yearbook, published by the U.S. Bureau of Mines through 1994 and by the U.S. Geological Survey from 1994 to present). The emission factors used, 0.059 ton CO_2-C/1 ton limestone and 0.064 ton CO_2-C/1 ton dolomite, are lower than the default IPCC emission factors because they account for a portion of limestone that may leach through soils and travel through waterways to the ocean (West & McBride 2005). The methodology summarized above is described in detail in Chapter 7 of the U.S. GHG Inventory (EPA 2010).

3.7 Uncertainty in Estimating Carbon Stock Changes in Agricultural Soils

Uncertainty was calculated separately for the Tier 3 and Tier 2 approaches used to estimate CO_2 fluxes. The methodologies summarized below are described in detail in Chapter 7 and Annex 3.13 of the U.S. GHG Inventory (EPA 2010).

3.7.1 Tier 3 Approach for Cropped Mineral Soils Simulated by CENTURY

As estimated by the CENTURY model, mineral soils on which major crops are grown sequestered 42 Tg CO_2 eq. in 2008 with a 95% confidence interval of +/- 64%. This uncertainty has three components: Monte Carlo approach to address uncertainties in CENTURY model inputs, an empirical approach to address structural uncertainty inherent in the model, and scaling uncertainty associated the NRI survey data. For model input uncertainty, probability distribution functions were developed for fertilizer rates, manure application, and tillage practices. A Monte Carlo analysis was conducted with 100 iterations in which input values were randomly drawn from the probability density functions to simulate the soil carbon stocks for each NRI cluster of points using CENTURY. An empirically based estimator was used to assess model structural error. This estimator was derived from a linear effects mixing model analysis of comparisons between modeled soil carbon stocks and measurements from 45 long-term experiments with over 800 treatments representing a variety of cropping, fertilizer, and tillage management practices (Ogle et al. 2006a). The model included variables that accounted for significant biases (alpha

level of 0.05) in CENTURY model estimates. For each carbon stock estimate from the Monte Carlo simulations,
the structural uncertainty estimator was applied to adjust the model output for bias and prediction error. Uncertainty in land use statistics from the NRI were incorporated based on the sampling variance of the cluster of NRI points.

3.7.2 Tier 2 Approach for Remaining Cropped Mineral Soils, Organic Soils, and Liming

The CENTURY model has not been adequately tested with organic soils and soils used to grow non-major crop types (e.g., commodity crops, vineyards, fruit and nut trees) so an IPCC Tier 2 methodology was used for these soils. As estimated by Tier 2 methodology, mineral soils for non-major crops lost ~1 Tg CO_2 eq. in 2008 with a 95% confidence interval of –380% and +377% and organic soils emitted 30 Tg CO_2 eq. in 2008 with a 95% confidence interval of –39% and +31%. A Monte Carlo approach was used to simulate a range of values with 50,000 iterations by selecting values from probability distribution functions (Ogle et al. 2003). For mineral soils, probability distribution functions were derived from a synthesis of 91 published studies that addressed the impact of land management on soil carbon stock changes. For organic soils, probability distribution functions for emission factors were derived from a synthesis of 10 studies and combined with uncertainties in the NRI land use data for organic soils.

As estimated by Tier 2 methodology, liming of soils led to emissions of ~4.0 Tg CO_2 eq. in 2008 with a 95% confidence interval of –97% and +102%. Uncertainty in the emissions factors and uncertainty in data for agricultural use of limestone and dolomite were included in the analysis.

3.7.3 Combined Uncertainties

Uncertainties for the above components were combined using simple error propagation (IPCC 2006). That is, the combined uncertainty was calculated by taking the square root of the sum of the squares of the standard deviations of the components. The combined 95% confidence interval for CO_2 storage in cropped soils in 2008 ranged from -38 to 20 Tg CO_2 eq. around the estimate of -8 Tg CO_2 eq. (Table 3-1).

3.7.4 Changes Compared to the 2nd Edition of the USDA GHG Report

There were important changes in land classification data that effected C stock change estimates. Data from the USDA National Resources Inventory (NRI) are used to classify land use and management practices. In previous inventories, NRI data were collected in 5-year increments, and the last available year was 1997. Availability of new annual data extended the time series of activity data beyond 1997 to 2003. Also, each NRI point was simulated separately instead of simulating clusters of points that had common cropping rotation histories, and more exact cropping histories were simulated instead of generalized cropping rotations. Overall, these changes resulted in an average annual decline in soil C sequestration in mineral soils of close to 20 Tg CO_2 eq. for the reporting period compared to the previous Inventory. Uncertainties are also higher because soil C stock changes were estimated for each year from new annual NRI data instead of averaging over 5-year periods.

In addition, annual C flux estimates for mineral soils between 1990 and 2008 were adjusted to account for additional C stock changes associated with sewage sludge amendments using a Tier 2 method provided in IPCC (2003, 2006), which utilizes U.S.-specific C loss rates (Ogle et al. 2003) rather than default IPCC rates. Estimates of the amounts of sewage sludge N applied to agricultural land were derived from national data on sewage sludge generation, disposition, and nitrogen content. Total sewage sludge generation data for 1988, 1996, and 1998, in dry mass units, were obtained from an EPA report (EPA 1999), and estimates for 2004 were obtained from an independent national biosolids survey (NEBRA 2007). These values were linearly interpolated to estimate values for the intervening years. The stock change rate is based on country-specific factors and the IPCC default method.

3.8 Mitigation of CO_2 Emissions

Currently, cropped soils in the U.S. are estimated to be storing carbon at the gross rate of approximately 43 Tg CO_2 and a net rate of ~8 Tg CO_2 per year. However, the potential to store carbon is thought to be much higher (e.g., Sperow et al. (2003) estimated a potential of 220 – 255 Tg CO_2 per year). Strategies to increase soil C stocks include: reduction in tillage intensity, reduced cropping of organic soils, reduced summer fallow, planting non-growing season cover crops, increased land in CRP, and increased use of hay or pasture in crop rotations. Organic soils provide an opportunity to mitigate emissions because they make up less than 1% of total cropped land in the U.S., but are a source of 30 Tg CO_2 per year (Table 3-7). Summer fallow tends to decrease soil carbon because during a large part of the growing season plants are not present to provide carbon inputs but decomposition of soil carbon by microbes continues. Cropped land converted to CRP stores carbon because the land is not cultivated and trees or grasses are planted to provide carbon inputs. Including hay or pasture in rotations also increases carbon inputs, and carbon losses are lower because the land is not tilled during the hay or pasture phase of the rotation. We do not quantitatively estimate mitigation potential for this report because no recent nationwide analyses have been conducted.

Recent data suggest that a large portion of the cropped land in the U.S. is currently under reduced or no till cultivation (Maps 3-4, 3-5), thus the potential for further soil carbon gains by reducing tillage intensity may be limited. However, reduced tillage intensity does imply reduced on-farm energy consumption and lower CO_2 emissions. Similarly, the potential to convert cropland to idle CRP land is limited because the demand for biofuel feedstocks has incentivized keeping lands in production. Currently, about one-third of the corn crop is used for ethanol production, and the amount of cropland dedicated to biofuel feedstock production is expected to continue to increase as the nation moves towards the goal outlined in the Energy Independence and Security Act of 2007 to increase domestic ethanol production from the current level of ~11 billion to 36 billion gallons by 2022. A large portion of future biofuel feedstocks are expected to be supplied by perennial crops which can increase soil C stocks, but no national analyses to quantify this potential have yet been published.

Chapter 4: Carbon Stocks & Stock Changes in U.S. Forests

4.1 Summary

Forest ecosystems, urban trees, and forest products represent significant carbon sinks in the United States, offsetting approximately 12.7% of total U.S. greenhouse gas emissions. The net amount of carbon stored—that is, annual incremental change—by forests during 2008 in the United States is an estimated 704 and 88 Tg CO2 eq. for forest ecosystems and harvested wood products (HWP), respectively. Net forest system (ecosystems plus HWP) total sequestration in 2008 was estimated to be 792 Tg CO2 eq., with a 95% confidence interval of 935 to 651 Tg CO2 eq. (Table 4-1). Compared to 1990, CO2 sequestered by forest systems in 2008 was about 8% greater (Table 4-2). Although the net effect was zero, an additional 194 Tg CO_2 eq. was sequestered by trees, but harvested and burned to produce energy. Urban trees also sequestered carbon, about 94 Tg CO_2 eq. in 2008. Current total carbon stocks in forest ecosystems of the conterminous United States are

Table 4-1 Forest Carbon Stock Change Estimates and Uncertainty Intervals for 2008

	Estimate	95% Confidence Interval
Source		*Tg CO_2 eq.*
Forest	(704)	(846) to (567)
Harvested Wood	(88)	(110) to (67)
Total	**(792)**	**(935) to (651)**

Note: Parentheses indicate net sequestration. Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Table 4-2 Carbon Stocks and Annual Change for Forest and Wood Pools and Forest Area, 1990, 1995, 2000, 2005-2008[1]

	1990	1995	2000	2005	2006	2007	2008
Annual Change				*Tg CO_2 eq. yr^{-1}*			
Forest	**(598)**	**(574)**	**(355)**	**(701)**	**(704)**	**(704)**	**(704)**
Aboveground Biomass	(378)	(398)	(309)	(397)	(397)	(397)	(397)
Belowground Biomass	(74)	(79)	(62)	(79)	(79)	(79)	(79)
Dead Wood	(29)	(31)	(16)	(23)	(26)	(26)	(26)
Litter	(47)	(28)	3	(56)	(56)	(56)	(56)
Soil Organic Carbon[2]	(70)	(37)	29	(146)	(146)	(146)	(146)
Harvested Wood	**(132)**	**(118)**	**(113)**	**(105)**	**(109)**	**(103)**	**(88)**
Wood Products	(65)	(55)	(47)	(45)	(45)	(39)	(24)
Landfilled Wood	(67)	(63)	(66)	(60)	(63)	(64)	(64)
Total	**(730)**	**(693)**	**(468)**	**(807)**	**(812)**	**(807)**	**(792)**
Carbon Stock				*Tg CO_2 eq.*			
Forest	**155,981**	**158,884**	**161,235**	**164,126**	**164,827**	**165,531**	**166,235**
Aboveground Biomass	55,098	57,016	58,775	60,606	61,003	61,400	61,797
Belowground Biomass	10,948	11,328	11,677	12,041	12,120	12,199	12,278
Dead Wood	10,814	10,964	11,093	11,193	11,216	11,242	11,269
Litter	17,436	17,644	17,715	17,892	17,948	18,004	18,060
Soil Organic Carbon	61,685	61,932	61,974	62,394	62,539	62,685	62,831
Harvested Wood	**6,817**	**7,440**	**8,021**	**8,525**	**8,631**	**8,739**	**8,842**
Wood Products	4,514	4,807	5,069	5,264	5,309	5,354	5,393
Landfilled Wood	2,303	2,633	2,952	3,262	3,322	3,385	3,449
Total	**162,798**	**166,323**	**169,256**	**172,651**	**173,458**	**174,270**	**175,077**
				1,000 ha			
Forest Area	**267,986**	**271,194**	**273,767**	**276,796**	**277,536**	**278,276**	**279,016**

Note: Parentheses indicate net sequestration. Tg CO2 eq. is teragrams carbon dioxide equivalent.

[1]Based on interpolation and extrapolation after aggregating plot-level data to state totals according to Smith et al. (2010).

[2]Soil carbon does not include effects of land use history.

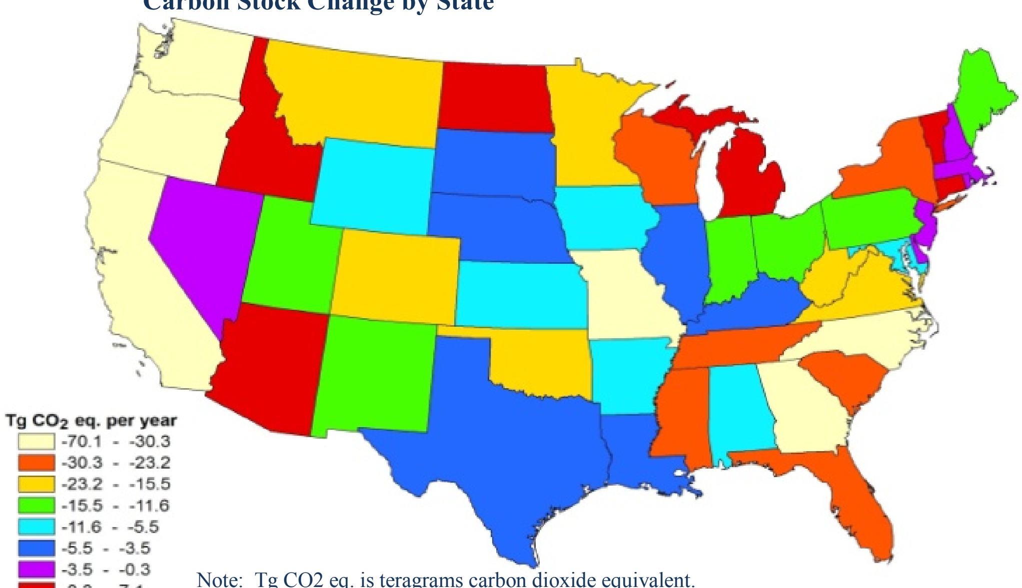
about 167 Pg CO2 eq. (Table 4-2, Pg=1,000 Tg).

Periodic summary statistics on forestland in the conterminous United States indicate about a 3% increase in area between the compilation years 1987 to 2007, that is, about 9 million additional hectares (Smith et al. 2009). In addition to the net accumulation of carbon in harvested wood pools, sequestration is a reflection of net forest growth and increasing forest area over this period. Generally, the largest stocks and net annual changes are in biomass carbon.

Carbon sequestration rates for forests and harvested wood products are greatest in California, followed by Missouri, Georgia, Washington, Oregon, North Carolina, Wisconsin, and Mississippi (Map 4-1). Only six States are emitting more carbon than they are sequestering. The distribution of forestland in the conterminous United States is illustrated in Map 4-2; carbon stock and change summaries provided below are according to the 10- or 4-region sets specified in Map 4-3. Among the four regions, total carbon stocks and net annual change (sequestration) are greatest in the North. However, stock and change are greatest in the Pacific Coast region when expressed on a per-hectare basis (see Table C-1 for details of this summary). Hardwood forest type groups in the East formed the largest stock of carbon in biomass; this was about 27 Pg CO_2 eq. in comparison to about 15 Pg CO_2 eq. in softwood and mixed type groups in the East (Table 4-3). Softwood type groups in the West included about 25 Pg CO_2 eq. in biomass, whereas hardwood type groups accounted for about 4 Pg CO_2 eq.

Forestlands of the United States constitute 33% (304 million hectares) of total U.S. land area. These forestlands are surveyed by the USDA Forest Service, Forest Inventory and Analysis (FIA) program. A large proportion of these forests are managed for timber production. About 75% of forestland, 277 million hectares are classified as timberland, meaning they meet minimum levels of productivity and are available for timber harvest. Effects of management and land use change are implicitly part of the forest survey and are thus reflected in carbon stocks and stock changes. This chapter summarizes carbon stocks and stock changes on an average 273 million hectares located in the conterminous 48 States and coastal Alaska. Summaries of information included in this chapter represent updates of inventories and carbon estimations relative to the national

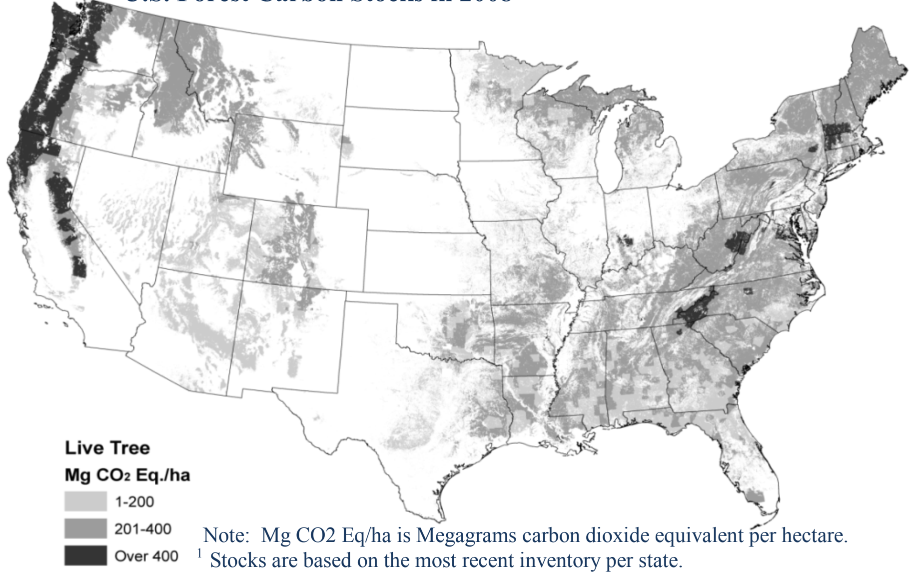
Map 4-1
Carbon Stock Change by State[1]



Note: Tg CO2 eq. is teragrams carbon dioxide equivalent.
[1] Net change values are model estimates for 2008 according to Smith et al. (2010)

forest carbon budgets reported in the second edition of the USDA Greenhouse Gas Inventory (Smith & Heath 2008). Estimates of stocks and net annual stock change for carbon on forestlands and in harvested wood products for the conterminous United States presented here expand on the information reported for forestlands in Heath et al. (in press) and Chapter 7 of the most recent U.S. GHG Inventory (EPA 2010), and are consistent with reporting recommendations of the Intergovernmental Panel on Climate Change (IPCC) Good Practice Guidance for Land Use, Land-Use Change, and Forestry (Penman et al. 2003). The summary tables provided in this chapter and in appendix C provide additional detail by summarizing data according to forest types, ownerships, or other classifications.

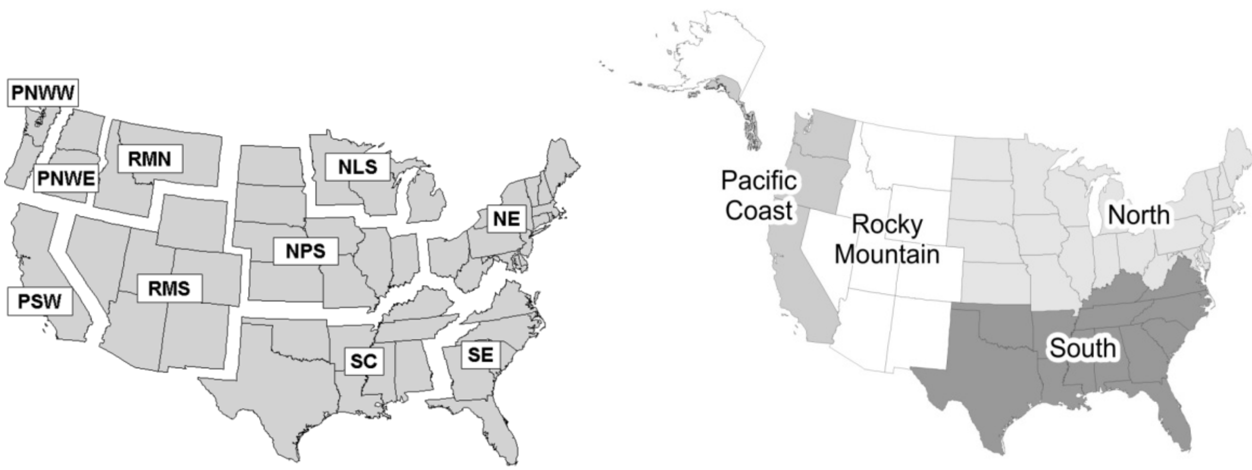
Map 4-2
U.S. Forest Carbon Stocks in 2008[1]



Note: Mg CO2 Eq/ha is Megagrams carbon dioxide equivalent per hectare.
[1] Stocks are based on the most recent inventory per state.

Although annual estimates are available beginning in 1990, we present estimates for a logical subset of years (Table 4.3). The post-2000 large increase in sequestration (Table 4-2) is due

Map 4-3
Regions Used for Carbon Stock and Stock-Change Summaries[1]



[1] Regions used for 10- or 4-region carbon summaries are: Pacific Northwest, West (PNWW); Pacific Northwest, East (PNWE); Pacific Southwest (PSW); Rocky Mountain, North (RMN); Rocky Mountain, South (RMS); Northern Prairie States (NPS); Northern Lake States (NLS); Northeast (NE); South Central (SC); and Southeast (SE). Note that regions are merged for some tables, these combinations include: PNWW, PNWE, PSW, and coastal Alaska as Pacific Coast; RMN and RMS as Rocky Mountain; NLS, NPS, and NE as North; and SC and SE as South. Pacific Coast and Rocky Mountain are collectively called West, and North and South are collectively referred to as East.

mainly to additional increases in the East, mostly in the North. The main factor in the increase is the relatively steady increase in forest area in many states; stocks follow this area trend. There is an additional increase in soil organic carbon (SOC) beyond that expected from increases in forest area. This additional SOC change is a function of forest type groups. The total effect in SOC net change over the entire interval (1990-2008) suggests there was a slight shift to low specific SOC density forest types in the 1990s relative to earlier and later inventories (see Annex 3.12 of EPA (2010) for SOC associated with forest types).

Table 4-3 Forest Area, Carbon Stocks, and Net Annual Stock Change by Forest Type Group[1]

		Carbon Stocks			Net Annual Stock Change		
	Forest Area	Biomass	Dead Plant Matter	SOC[2]	Biomass	Dead Plant Matter	Per Hectare
Forest Type	*1,000 ha*		*Tg CO_2 eq.*		*Tg CO_2 eq. Yr^{-1}*		*kg CO_2/ha*
East	**176,735**	**43,766**	**12,592**	**45,666**	**(321)**	**(35)**	**(18,820)**
Aspen/Birch	6,963	1,293	453	3,353	10.8	3.4	2,041
Elm/Ash/Cottonwood	8,805	2,116	488	2,963	(0.7)	1.7	113
Loblolly/Shortleaf Pine	22,901	4,757	1,447	4,622	(84.2)	(18.3)	(4,476)
Longleaf/Slash Pine	5,320	842	314	1,907	(9.0)	(1.2)	(1,919)
Maple/Beech/Birch	19,912	6,729	2,855	6,116	(28.8)	0.5	(1,420)
Oak/Gum/Cypress	9,716	3,177	569	3,606	5.3	3.2	867
Oak/Hickory	63,590	18,414	3,613	11,218	(240.8)	(35.1)	(4,339)
Oak/Pine	11,800	2,724	856	2,326	33.3	11.1	3,765
Pinyon/Juniper	3,575	245	190	499	n/a	n/a	n/a
Spruce/Fir	6,129	1,242	956	4,111	2.8	(0.3)	415
White/Red/Jack Pine	4,170	1,376	362	1,440	4.4	4.1	2,033
Woodland Hardwoods	7,441	304	158	1,773	n/a	n/a	n/a
Other Hardwood Type Groups	850	141	34	357	(7.9)	(1.5)	(11,022)
Other Softwood Type Groups	1,902	319	170	349	(6.2)	(3.0)	(4,879)
Nonstocked	3,661	86	128	1,024	n/a	n/a	n/a
West	**99,971**	**28,739**	**16,536**	**16,878**	**(11)**	**(12)**	**(9,784)**
Alder/Maple	1,353	503	162	556	n/a	n/a	n/a
Aspen/Birch	3,506	789	502	724	n/a	n/a	n/a
California Mixed Conifer	3,167	1,817	823	578	n/a	n/a	n/a
Douglas Fir	15,785	7,229	3,490	3,831	n/a	n/a	n/a
Fir/Spruce/Mountain Hemlock	14,756	5,929	3,541	2,560	n/a	n/a	n/a
Hemlock/Sitka Spruce	5,011	3,280	1,547	2,018	n/a	n/a	n/a
Lodgepole Pine	6,467	1,527	865	873	n/a	n/a	n/a
Other Western Softwoods	3,425	488	549	670	n/a	n/a	n/a
Pinyon/Juniper	19,890	1,995	1,616	1,493	(3.7)	(3.1)	(340)
Ponderosa Pine	9,313	2,008	1,128	1,236	n/a	n/a	n/a
Redwood	285	262	112	56	n/a	n/a	n/a
Spruce/Fir	383	39	55	87	n/a	n/a	n/a
Tanoak/Laurel	1,101	594	176	218	n/a	n/a	n/a
Western Larch	697	190	135	95	(3.1)	(1.7)	(6,937)
Western Oak	4,304	1,219	570	504	n/a	n/a	n/a
Western White Pine	112	31	17	19	n/a	n/a	n/a
Woodland Hardwoods	4,583	396	552	470	(4.0)	(7.4)	(2,506.5)
Other Hardwood Type Groups	1,163	315	135	249	n/a	n/a	n/a
Nonstocked	4,669	128	561	641	n/a	n/a	n/a
Total	**276,706**	**72,505**	**29,127**	**62,544**	**(332)**	**(48)**	**(28,603)**

[1]Net change is determined from the two most recent inventories for all forests. Stock change does not include soil carbon changes. Stocks and area are based on the most recent inventory per state. Tg CO2 eq. is teragrams carbon dioxide equivalent. Kg CO_2 is Kilograms carbon dioxide.
[2]SOC (soil organic carbon) does not include effects of past land use history.
Note: "n/a" Indicates not available because large area of comparable forest type groups is not defined for two surveys within the Forest Inventory Analysis database (FIADB v4.0). Totals over these columns should be interpreted accordingly.
Note: Parentheses indicate net sequestration. Other Hardwood Type Groups and Other Softwood Type Groups represent aggregates of minor type groups. However, "Other Western Softwoods" is a specific type group within the FIADB.

The transition from FIA's older periodic inventories to the current annualized inventories does not affect the estimates in an identifiable way. Both types of surveys provide estimates of quantities of forest land and forest characteristics on that land; we have identified and corrected for known differences occurring over time (Smith et al. 2010, EPA 2010).

4.2 Concepts and Conventions

For reporting purposes, carbon estimates in forest ecosystems are allocated to the following pools (Penman et al. 2003):

- Aboveground biomass, which includes all living biomass above the soil including stem, stump, branches, bark, seeds, and foliage. This category includes not only live trees, but live understory.
- Belowground biomass, which includes all living biomass of coarse living roots greater than 2 mm diameter.
- Dead wood, which includes all non-living woody biomass either standing, lying on the ground (but not including litter), or in the soil.
- Litter, which includes the litter, fumic, and humic layers, and all non-living biomass with a diameter less than 7.5 cm at transect intersection, lying on the ground.
- Soil organic carbon (SOC), all organic material, including fine roots, in soil to a depth of 1 meter but excluding the coarse roots of the belowground pools.

The two harvested wood products carbon pools are:

- Harvested wood products in use.
- Harvested wood products in solid waste disposal sites (SWDS).

Continuous, regular annual surveys are not available over the entire time period of interest for each state; therefore, estimates for non-survey years were derived by interpolation between known data points. Survey years vary by state and the list of survey years and data can be found in Table 2 in Smith et al. (2010). Thus, the national estimates in Table 4-2 are a composite of individual state surveys, broken out in more detail in Appendix Table C-1. The same process applies to forest area for each year – annual data are not available throughout the interval so annualized average information between inventory years is presented here (Smith et al. 2010).

4.3 Carbon Stocks and Stock Changes by Forest Type, Region, and Ownership

Total forest ecosystem areas, carbon stocks, and net annual stock change according to forest type group are listed in Table 4-3. Minor type groups in the East and West are pooled, for example, tropical and exotic hardwood groups in both regions. Carbon classifications in this table are for biomass, nonliving plant mass, and soil organic carbon. Biomass includes live trees plus live understory vegetation. Non-living plant mass includes standing dead trees, down dead wood, and the forest floor. Carbon estimates include aboveground and belowground components.

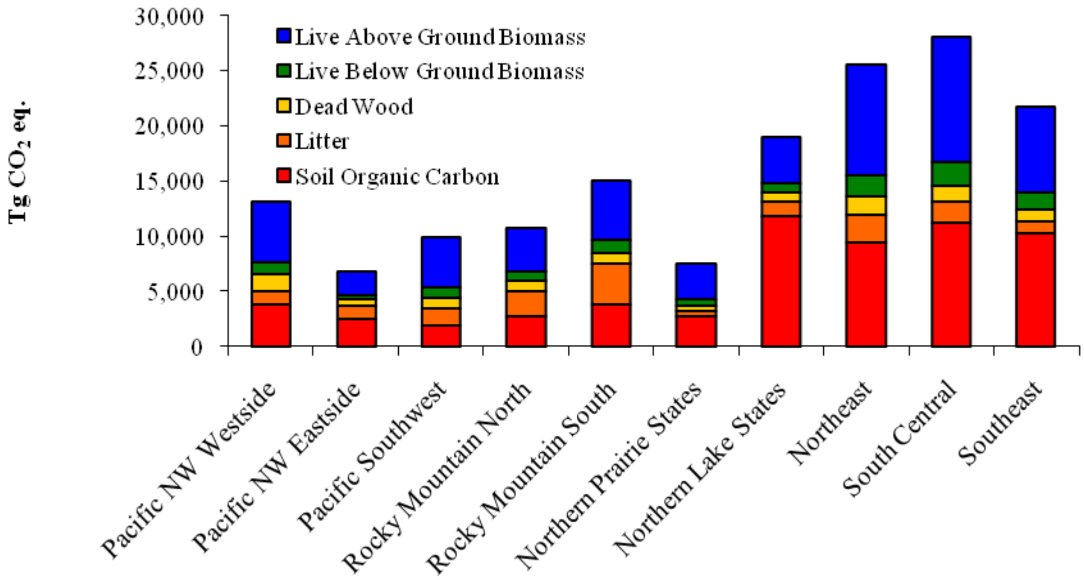
Estimates of stock change according to forest type group were developed by subdividing the state or sub-state classifications according to forest type group (USDA FS 2010) before calculating annualized stock or stock change. Note that not all forestland includes at least two surveys in the current inventory format, which limits the estimates of change available in Table 4-3. Thus, change calculated for selected subsets does not necessarily add to totals calculated as more aggregate stocks.

Regional summaries were developed for the regions indicated in Map 4-3; the 10-region classifications are used in Figures 4-1 and 4-2, while the 4-region set is used for additional tables in the appendix. Total forest ecosystem carbon stocks are generally greater in eastern regions than in the West (Figure 4-1a). This is in contrast to regional average values for carbon density, which are greater in the West than in the East (Figure 4-1b). Mass of carbon per unit area is greatest in the Pacific Northwest-Westside and the Northern Lake States due to large pools of biomass and SOC, respectively. The most notable regional features in ecosystem pool carbon density are: greater carbon in biomass in the Pacific Northwest-Westside; greater SOC pools in northern regions; and smaller pools of down dead wood and forest floor in the South. Net annual stock changes are shown in Figure 4-2, which includes estimated changes in harvested wood product pools.

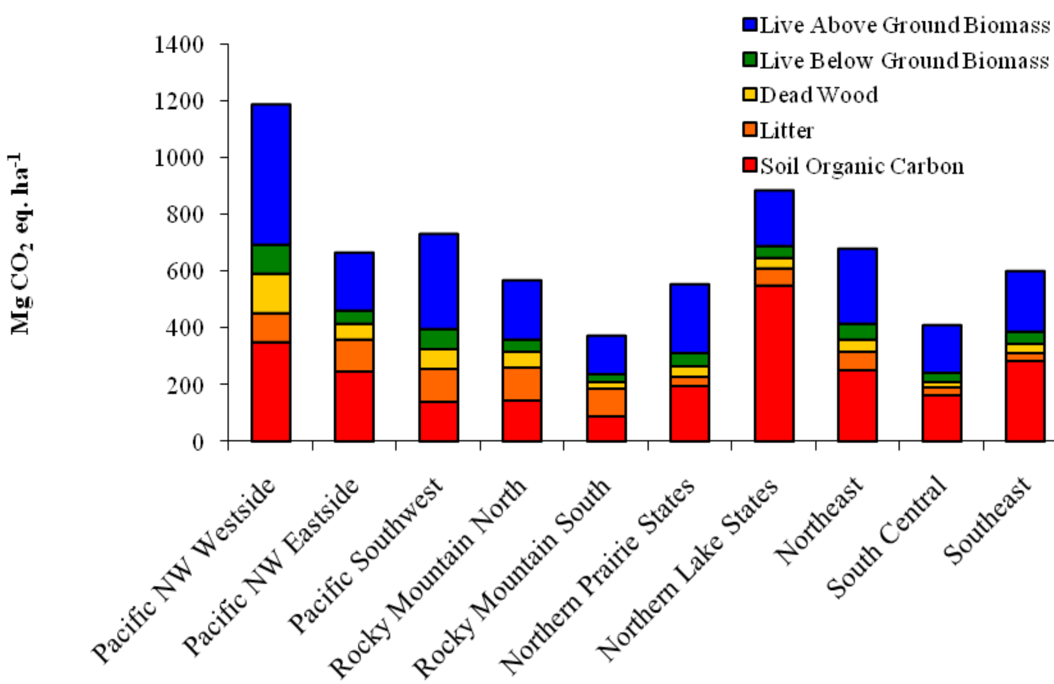
Forestland in the conterminous United States is distributed throughout the 48 States. Carbon density of live trees, both aboveground and belowground, is shown in Map 4-2, which illustrates both the spatial distribution of forest ecosystem carbon and average carbon density over the lower 48 States. Large areas of high live-tree carbon density include the Pacific Coast states and the Appalachian Mountains. This map is based on the most recent inventory data available per state. (State-wide summaries of total forest area and non-soil ecosystem carbon stock are presented in Appendix Table C-1.) This table also includes net change for area, non-soil ecosystem carbon stock, and stock of carbon in harvested wood products for 2008. Carbon stock change in harvested wood is allocated according to total roundwood removals per state reported for nominal year 2006 in the Timber Product Output tables available at USDA FS (2010). Calculated values for net annual change in forest ecosystem carbon reflect estimated carbon densities and forest areas reported in the two most recent surveys per state.

Figure 4-1

a) **Forest Ecosystem Carbon Stocks**



b) **Forest Ecosystem Average Stock Density**[1]



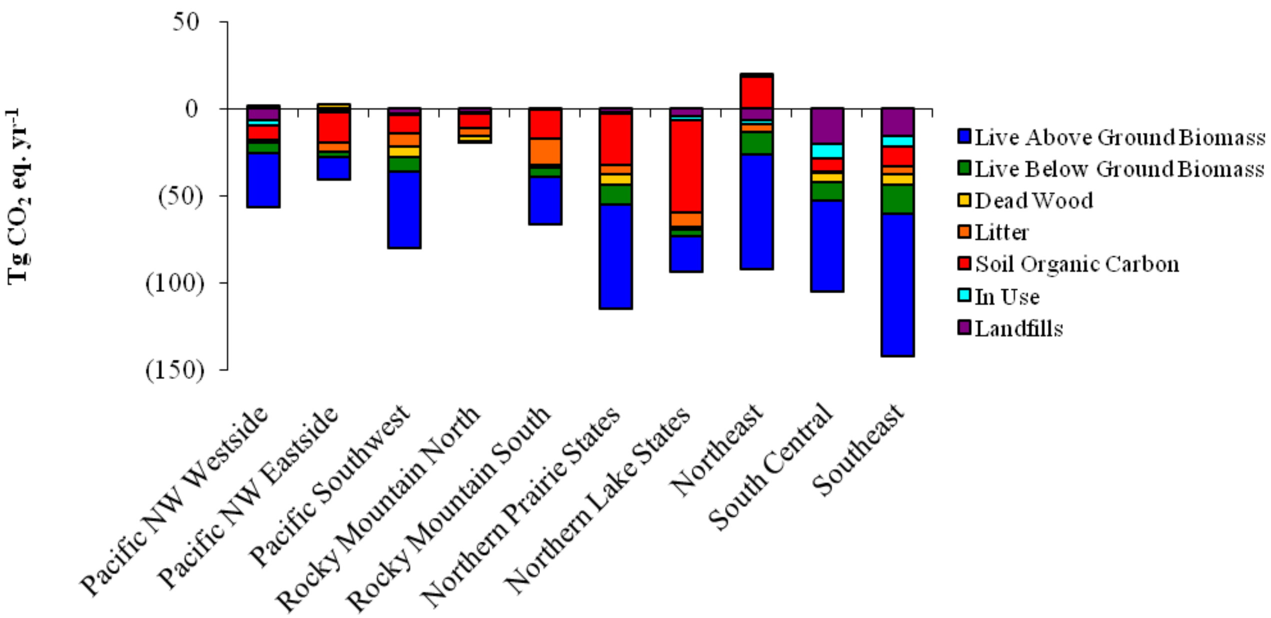
Note: Soil carbon does not include effects of land use history.

[1] Based on plot-level data from the most recent inventory available per state.

Tg CO_2 eq. is teragrams carbon dioxide equivalent; Mg CO_2 eq. ha^{-1} is Megagrams carbon dioxide equivalent per hectare.

Estimates of net annual change calculated as the difference between two successive inventories are sensitive to changes in forestland over the interval as well as changes in average carbon density. Even small differences in carbon density can contribute to large differences if the change is applied to large areas. Whether change in area or density is the controlling factor is dependent on the situation (Smith and Heath 2010). Most estimates of net ecosystem carbon change provided in Tables 4-2 and 4-3, Figure 4-2, and Appendix Table C-1 correspond well to changes in forest area. That is, net gains in forest carbon are most often accompanied by increases in forestland and vice versa. There are exceptions, and most of these involve net gains in forest carbon (negative flux) despite decreases in area. This is the case in Table 4-3 for Eastern Maple Beech Birch forests which are decreasing in overall area (data not shown), yet total carbon stock in biomass is increasing. Similarly, Appendix Table C-1 shows the pattern of

Figure 4-2
Net Annual Forest Carbon Stock Change[1]



[1] Based on interpolation and extrapolation after aggregating plot-level data to state totals.
Parentheses indicate a net sequestration. Soil carbon does not include effects of land use history.
Tg CO_2 eq. is teragrams carbon dioxide equivalent.

carbon stock trend counter to forest area trend in 16 of the lower 48 States listed. The two instances of net carbon loss accompanying area gains involve relatively low rates of area change (0.2% or less).

Additional tabular summaries of forest ecosystem carbon stocks are provided in Appendix Tables C-2 through C-5. The distribution of carbon stocks among forest age classes is shown in Appendix Table C-2 for privately owned and Appendix Table C-3 for publicly owned forests. The tables illustrate that the greater proportion of forest carbon stocks in the East is under private ownership whereas the greater proportion in the West is under public ownership. Distributions according to age are shifted toward older forests on public lands; this is the case for all four regions but is more apparent in the West. Similarly, distribution according to stand size class (Appendix Table C-4) shows a greater proportion in larger size class stands in the West.

Patterns of carbon stocks among forest types and ownerships are presented by forest ecosystem pools (excluding soils) in Appendix Table C-5. Ownership is classified as public or private for timberlands (forests of minimum productivity and available for harvesting). The remaining forestland, both public and private, is either reserved from harvesting or is considered less productive (and thus probably not managed for commercial wood products). The net annual stock change corresponding to Appendix Table C-5 is provided in Appendix Table C-6. Note that Appendix Table C-6 is affected by the same data limitations as discussed above for Table 4-3. For more information about forest inventory variables such as forest classifications of ownership, productivity, forest type, and stand size class, see Smith et al. (2009) and USDA Forest Service (2010).

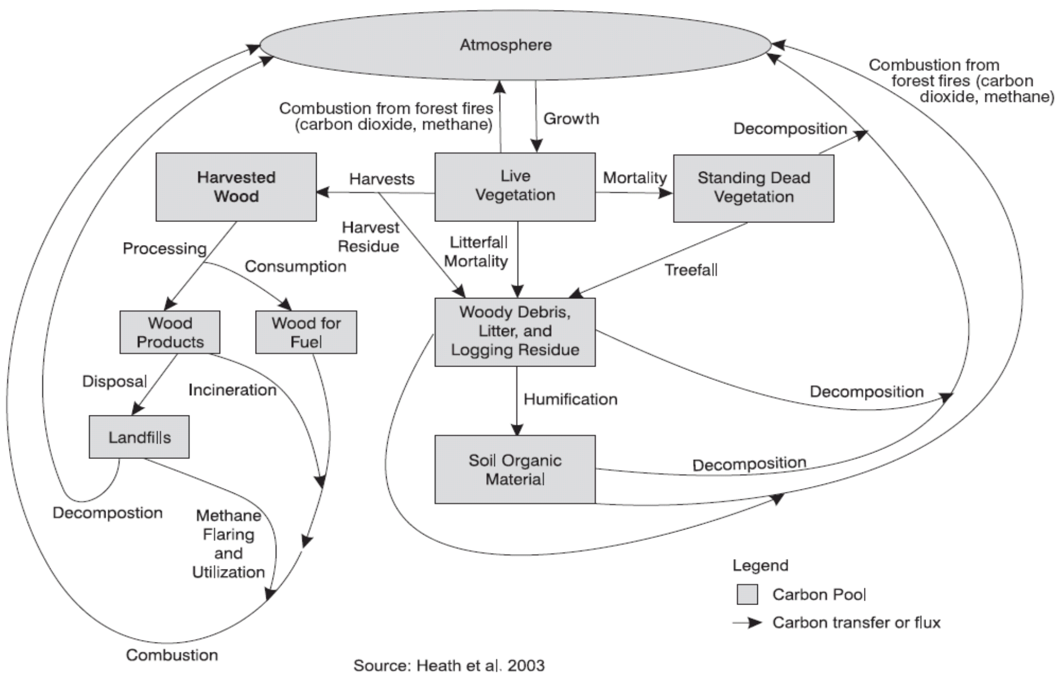
A large proportion of non-forest trees in the United States is in urban areas – approximately 3% of total tree cover in the conterminous United States, with notable urban expansion projected in the future (Nowak and Walton 2005). Advances in design and deployment of trees in urban environments can provide significant fossil fuel savings for heating and cooling through microclimate management (Dwyer et al. 2000). Development of urban tree waste management and recycling processes and systems would reduce emissions and increase sequestration opportunities. Methods have been developed for estimating carbon sequestration rates for urban trees of the United States (Nowak & Crane 2002). Net flux of carbon into urban trees for 2008 is estimated to be -94 Tg CO2 eq. per year (EPA 2010).

4.4 Mechanisms of Carbon Transfer

Carbon sequestration is a function of the continuous exchange of carbon dioxide between forest ecosystems and the atmosphere, which is illustrated by Figure 4-3. Forest carbon balance also includes some non-CO2 emissions, but the majority of exchange is in terms of CO2, which is the focus of this chapter. Tree growth results in the net accumulation of CO2 in forests (removal from the atmosphere), whereas other processes such as respiration, decomposition, or combustion emit CO2 to the atmosphere. Photosynthesis provides the energy for the conversion of carbon dioxide to organic carbon; this assimilation of CO2 by trees most often exceeds any simultaneous losses through respiration, resulting in net tree growth. Forests convert much of the accumulated carbon to wood, which stores carbon and energy. Processes that control the fate of wood grown in a forest largely determine the subsequent loss of CO2 to the atmosphere. Mortality and disturbance add to the pools of down dead wood and forest floor, which are subject to decay. Carbon can also be removed from forest ecosystems through runoff or leaching through soil. Mechanisms of relatively rapid carbon loss from specific forestlands include disturbances such as fire or the harvest of wood. However, a portion of the carbon in harvested wood is not immediately returned to the atmosphere, rather it is retained in wood products. Once in a product pool, the carbon is emitted as CO2 over time through combustion or decay of the wood product. Net release of carbon from wood products can vary considerably depending on the product, its end use, and the means of disposal (Heath et al. 1996, Smith et al. 2006, Skog 2008).

Forest management affects carbon stocks and stock changes by controlling mechanisms associated with carbon gain and loss (Houghton & Hackler 2000, Johnson & Curtis 2001). Forest management can be defined as activities involving the regeneration, tending, protection, harvest, and utilization of forest resources to meet goals defined by the forestland owner. Management often focuses on more than one outcome and can vary by forest ecosystem, landowner objectives, and economic possibilities. Example goals, or expected outcomes, of management include productivity and resource conservation. Relatively passive management may include tree harvest and removal, followed by natural regeneration, or riparian area management such as consciously retaining a buffer strip of trees along a watercourse. Intensive management may consist of site preparation, improved stocking, species conversion, planting genetically improved

Figure 4-3. Summary diagram of forest carbon stocks and carbon transfer among stocks



stock, application of pesticides or fertilizer, and improvement cuttings such as thinning or pre-commercial thinning.

Increased net carbon sequestration is generally associated with forest systems under improved forest management practices, although some practices may reduce carbon storage for a given site-age-type dynamic. Examples of improved management practices include afforestation, increased productivity, reduced conversion to non-forest uses, lengthened rotations in some systems, and increased proportion and retention of carbon in harvested wood products. Afforestation offers significant opportunities to capture and store carbon on lands that are not currently forested (Houghton & Goodale 2004, Woodbury et al. 2006). This is a particularly useful approach to increasing carbon sequestration for marginal agricultural lands. Similarly, reductions in conversion to non-forest land uses contribute to maintaining carbon stocks, particularly through the additional organic carbon storage in forest soils (Lal 2005). Sustainable short-rotation woody crops systems offer the opportunity to rapidly deploy new, faster growing genetic material, sequester carbon in the soil, add to the wood products pool, and provide energy feedstocks as fossil fuel offsets. Improvements in the management of wood products in use and in landfills provide a number of opportunities to reduce emissions and increase sequestration.

Continuing development of wood products can increase their use as substitutes for nonrenewable materials and extend their durability and thus expected lifespan (Perez-Garcia et al. 2005).

Some of the carbon in harvested wood products may remain sequestered for a long time before returning to the atmosphere, depending on the lifespan of the individual products. Emissions can occur from wood burned for energy, or from decay or burning of wood without energy capture (Figure 4-3). This distinction between the two paths for carbon emitted to the atmosphere is useful to assess potential displacement of other fuel sources. Average annual carbon emissions from harvested wood are estimated at 373 Tg CO2 eq. over the period 1990 through 2008 (EPA 2010, see Table A-220 of Annex 3.12). In 2008, it is estimated that 194 Tg CO_2 eq. was emitted by combustion of wood biomass to produce energy (EPA 2010). Net annual carbon sequestration via harvested wood, after accounting for these emissions, is presented in Table 4-2. Information about emissions released when processing or producing harvested wood carbon pools in the U.S. can be found in Heath et al. (2010).

4.5 Methods

Estimates of forest ecosystem carbon as reported here are based on the stock-change method, which uses forest inventory data to produce a series of successive carbon stock estimates for an individual state, for example (Penman et al. 2003, Smith et al. 2010). The FIA data consist of a series of annual partial surveys per state each year with re-measurements at 5- to 10-year intervals, depending on the state (USDA FS 2010). Carbon stocks for each forest classification, ecosystem carbon pool, and inventory are separately calculated and aggregated to total stocks for a specific year for each state. The term “survey” is used here to describe a complete inventory for a state, which is repeated at regular intervals. The inventories for some states are further divided into separate sub-state classifications for consistency in each consecutive series of carbon stocks. Net annual stock change (sometimes referred to as flux) is the difference between successive stocks divided by the interval of time between surveys. Carbon estimates for harvested wood products are based on a separate stock change method and input data that are not directly related to forest inventory data.

The overall goal in reporting these pools is to be as consistent as possible with: (1) the format and estimates provided in the previous USDA forest carbon inventory (Smith & Heath 2008); (2) current forest carbon estimates (EPA 2010, Heath et al. in press); and (3) the carbon estimation methods applied to the available inventory data. As a result, the sequence and identity of figures and tables describing forest carbon are similar to the previous inventory, but the estimates are updated to those in EPA (2010). Classifications, or groupings, of values within tables or figures have changed somewhat due to corresponding changes in forest inventories. Methods are summarized below, with additional details in EPA (2010), Skog (2008), and Smith et al. (2010).

Current forest survey data for the United States are available from the Forest Inventory and Analysis DataBase (FIADB), version 4.0 (USDA FS 2010). Surveys from the FIADB are supplemented with some older surveys from FIA Resources Planning Act Assessment (RPA) databases, which are periodic summaries of state inventories, along with older FIA tree-level data for some states. More complete information about FIA forest inventories is available on the Internet (USDA FS 2010). All FIADB surveys used for carbon stock estimates were obtained from the FIADB Web site on December 4, 2009. See Table 2 of Smith et al. (2010) for a list of the specific surveys used for the estimate provided here.

Carbon estimation factors (described below) are applied to the plot-level inventory data and summed to calculate carbon stocks for each survey of each state. Each survey is associated with an average year for field collection of data. Carbon stocks for each state or sub-state classification are assigned to those average years with net stock change based on the interval (in years) between the stocks. In this way, state-wide annualized estimates of ecosystem stock and change can be calculated and summed to U.S. totals as presented in EPA (2010) and Table 4-2. A similar approach was used to produce the additional estimates disaggregated by categories presented in the figures and tables.

Forest ecosystem carbon is estimated for each inventory plot as six separate pools: live tree, understory vegetation, standing dead tree, down dead wood, forest floor, and soil organic carbon. Live tree and understory are also allocated to aboveground and belowground portions. For each inventory summary in each state, each carbon pool is estimated using available conversion factors, which are generally based on empirical or process-based models (Birdsey & Heath 1995, Birdsey & Heath 2001, Heath et al. 2003, Smith et al. 2004). Coefficients are applied to the survey data at the scale of FIA inventory plots; the results are estimates of carbon density (Mg per hectare). These densities are then converted to CO2 equivalents. Live tree and understory carbon pools are combined as biomass in this inventory. Similarly, standing dead trees and down dead wood are combined as dead wood in this inventory. Definitions of forest floor and SOC correspond to litter and forest soils, respectively.

Tree carbon includes aboveground and belowground (coarse root) carbon mass of live trees. Separate estimates are made for whole-tree and aboveground-only biomass. Thus, the belowground portion is determined as the difference between the two estimates. Tree carbon estimates are based on equations in Jenkins et al. (2003) and are functions of tree species and diameter as well as forest type and region. Tree carbon in the RPA plots, which do not include individual tree data, are estimated from plot-level growing stock volume of live trees and equations based on Smith et al. (2003). Carbon mass of wood is 50% of dry weight (Eggleston et al. 2006). The minimum-sized tree included in the tree carbon pool data is one-inch diameter (2.54 cm) at breast height (1.3 meter). Understory vegetation is defined as all biomass of undergrowth plants in a forest, including woody shrubs and trees less than one-inch diameter, measured at breast height. We estimated that 10% of understory carbon mass is belowground. This general root-to-shoot ratio (0.11) is near the lower range of temperate forest values provided in Penman et al. (2003), and was selected based on two general assumptions: (1) ratios are likely to be lower for light-limited understory vegetation as compared with larger trees, and (2) a greater proportion of all root mass will be less than 2 mm diameter. Understory carbon density estimates are based on Birdsey (1996). Coefficients used for estimating understory or the volume-based standing dead tree carbon are presented in Annex 3.12 of EPA (2010).

Dead wood includes down dead wood and standing dead trees. Down dead wood is defined as pieces of dead wood greater than 7.5 cm diameter, at transect intersection, that are not attached to live or standing dead trees. Down dead wood includes stumps and roots of harvested trees. Ratio estimates of down dead wood to live tree biomass were developed by FORCARB2 simulations and applied at the plot level (Heath et al. in press). The standing dead tree carbon pool includes aboveground and belowground (coarse root) mass. Estimates are based on Smith et al. (2003) and are functions of plot-level growing stock volume of live trees, carbon density of live trees, forest type, and region. Coefficients used for estimating dead wood carbon are presented in Annex 3.12 of EPA (2010).

Estimates of forest floor and SOC are not based on carbon density of trees, but are functions of plot age, forest type, and region. Forest floor carbon is the pool of organic carbon (litter, duff, humus, and fine woody debris) above the mineral soil and includes woody fragments with diameters of up to 7.5 cm. Estimates are based on equations of Smith and Heath (2002) applied at the plot level. Forest floor and woody debris remaining after harvests are also included as part of calculations of forest ecosystem carbon pools. Estimates of SOC are based on the national STATSGO spatial database (USDA SCS 1991, USDA NRCS 2006) and the general approach described by Amichev & Galbraith (2004). In their procedure, SOC was calculated for the conterminous United States using the STATSGO database, and data gaps were filled by representative values from similar soils. The SOC estimates are based on region and forest type only. Links to region and forest type groups were developed with the assistance of the USDA Forest Service FIA Geospatial Service Center by overlaying FIA forest inventory plots on the soil carbon map. Historical land use change effects are currently not included in the estimate of the soil carbon pool (Johnson & Curtis 2001, Woodbury et al. 2006). That is, soil carbon for areas which were cleared and plowed at one time, and then reverted to forest, are probably still accruing soil carbon. However, we currently assume that all forests of a given forest type within a region have the same amount of SOC. The regional averages for SOC according to forest type group are included in Annex 3.12 of EPA (2010).

The tabular forest carbon summary values are based on a short sequence of calculations, these are: (1) determine carbon density for individual inventory plots; (2) identify the date (year) associated with each survey based on when data were collected; (3) sum total carbon within each state or sub-state classification for each survey to get carbon stock according to specific classification and year; and (4) linearly interpolate, or extrapolate, to determine annualized stocks and net stock change (Smith et al. 2010). In this way, carbon stocks are calculated separately for each state based on inventories available since 1990 and for the most recent inventory prior to 1990. With this method, stock and flux since the most recent survey are based on extrapolation. Thus, the annualized estimates for 2008 will not exactly match the most recent data per state. In the results presented in this chapter, all estimates of 2008 net stock change (or flux) are based on the difference between the two most recent surveys (extrapolated values). Most values for carbon stock or forest area are based on the most recent data available for each state; the only exception is the set of annualized stocks provided in Table 4-2.

Calculations of carbon in harvested wood products are completely separate from the ecosystem estimates because the datasets and methods are largely unrelated. These estimates focus on carbon in wood removed from the forest; logging residues are part of the ecosystem pools. Carbon in harvested wood that is either in products in use or in products discarded in solid waste disposal sites (SWDS) is based on the methods described in Skog (2008). Estimates were developed for years from 1910 onward based on historical data from the USDA Forest Service (USDA 1964, Ulrich 1989, Howard 2001), and historical data as implemented in the framework underlying the North American Pulp and Paper (NAPAP, Ince 1994), the Timber Assessment Market, and the Aggregate Timberland Assessment System Timber Inventory models (TAMM/ATLAS, Haynes 2003, Mills & Kincaid 1992). From these data on annual wood and paper production, the fate of carbon in harvested wood was tracked for each year from 1910 through 2008; this included the change in carbon stocks in wood products, in SWDS, and carbon emitted to the atmosphere. The carbon conversion factors and decay rates for harvested carbon removed from the forest are taken from Skog (2008). To account for imports and exports, the production approach is used, meaning that carbon in exported wood is counted as if it remained in the United States, and carbon in imported wood is not counted. The carbon stock changes presented in this chapter represent the net amounts of carbon that continue to be stored in a product pool. Allocation of the national estimates from EPA (2010) to regions or states is based on roundwood removals per state from the FIA Timber Products Output reports for RPA nominal year 2006 (see Other Reporting Tools at USDA FS 2010).

4.6 Major Changes Compared to Previous Inventories

The estimates provided in Table 4-2 reflect a substantial number of incremental changes in methods and data between EPA (2007) and EPA (2010) in terms of net stock change since 1990. The accumulation of newer inventory data for most states, including stocks for coastal (southern and eastern) Alaska and western Texas, affect totals and changes compared to previous inventories. Any modifications or adjustments in a survey are accompanied by corresponding modifications to related older data as necessary to assure consistency over time within each series. Reassessment of the definition of forestland resulted in reduction of grassland area in the U.S. because woodlands previously designated as grassland are now considered forest land, thus increasing the estimation of soil C stock changes in these areas (EPA 2010). However, re-defining forestland also led to the removal of low cover, lower productivity woodlands areas from the surveys (Smith et al. 2009), which were included in the previous USDA (2008) inventory. In addition, a few older western inventories were removed from the stock-change calculations to improve year-to-year consistency between successive stocks. Lastly, the estimation procedures for obtaining carbon stocks from inventory have changed (see Bechtold & Patterson 2005) as well as the approaches to selecting available inventory data. See Smith et al. (2010) for more discussion of how inventory data were used to develop the current 1990-2008 estimates. For comparison of the respective inventory sets, see Tables A-186 and A-207 of EPA (2007) and EPA (2010), respectively. On average, these changes increased carbon stock estimates by approximately 8 percent.

4.7 Uncertainty

Uncertainty about forest inventory data and the carbon conversion factors applied to the inventory contributes to overall uncertainty of the carbon estimates. Contributing components include errors in sampling or measurements as well as unknowns or errors in the largely empirical models used to develop the carbon factors applied at the plot level.

Sampling error is determined separately for each carbon pool according to Bechtold and Patterson (2005). Additional related errors in this use of inventory data are based on resolving a state's forest inventory to carbon stock for a defined forest area at a single point in time. Some small error is possible if surveys conducted over a multi-year period are averaged to a single year before calculating stock change. However, if significant portions of a state's forest inventory were sampled on a completely different schedule as was the case with some of the older inventories, then the error would increase. For this reason, stocks and stock changes were separately determined at sub-state levels, such as national forests, in some Western states (Smith et al. 2010). The potential for an additional error comes from the use of successive surveys and the need for consistent definition, identification and inclusion of all forestlands within a state. Small errors in carbon stocks are reflected in stock-change calculations; for example, if small areas or ownerships are omitted from one of a pair of successive surveys, then a portion of the resulting state-wide change is due to the apparent change in forestland. Such problems with definition or inclusion of forestlands can have significant effects on calculated net flux.

Uncertainty associated with the estimates of specific carbon stocks varies by carbon pool and forest type. Carbon in trees is relatively well defined, and information on errors in estimates (Jenkins et al. 2003) makes it possible to develop quantitative estimates of uncertainty. Relative errors in the estimates for other ecosystem carbon pools are greater; these carbon conversion factors are generally based on extrapolations of site-specific studies, which may not adequately represent regional averages. Additionally, representative data are not available for all forest types; this also increases uncertainty as substitutions are required. An important source of uncertainty is high variability and general lack of precision possible in assigning estimates of SOC. Soil carbon is a large pool, but it changes relatively slowly. There is limited information available for assessing soil carbon or the cumulative effects of land use change, which can amount to significant stock changes when summed over large forest areas (Woodbury et al. 2006).

A quantitative uncertainty analysis was developed for estimates of total carbon flux. The analysis incorporated the information from preliminary uncertainty analyses and estimates of uncertainty in the carbon conversion factors (Heath & Smith 2000, Smith & Heath 2001, Skog et al. 2004, Skog 2008). Additional details on the analysis are provided in Chapter 7 and Annex 3.12 of EPA (2010). The uncertainty analysis was performed using the IPCC-recommended Tier 2 uncertainty estimation methodology, that is, the Monte Carlo simulation technique. The 2008 95% confidence interval for forest carbon stock changes is -935 to -651 Tg CO2 eq. , with a mean sink of -792 Tg CO2 eq. (Table 4-1). The 95% confidence interval for forest ecosystem sequestration is -846 to -567 Tg CO2 eq., and is -110 to -67 Tg CO2 eq. for harvested wood products.

Chapter 5: Energy Use in Agriculture

5.1 Summary of Greenhouse Gas Emissions from Energy Use in Agriculture

Approximately 0.8 quadrillion btu of direct energy was used in agriculture in 2008, resulting in almost 72 Tg of CO_2 emissions (Table 5-1). The total energy consumption for all sectors in the U.S., including agriculture, was approximately 100.9 quadrillion btu, resulting in 5,572.8 Tg of CO_2 emissions (EPA 2010). Production agriculture's contribution to this total was very small at a little less than 1.3%. Within production agriculture, diesel fuel accounted for about 38% of CO_2 emissions and electricity contributed about another 38% of CO_2 emissions. Gasoline consumption accounted for about 11% of CO_2 emissions, while LP gas and natural gas accounted for about 7% and 5% respectively.

Table 5-1 Energy use and carbon dioxide emissions by fuel source on U.S. farms in 2008

Fuels	Energy consumed	Carbon content	Fraction oxidized	CO_2 emissions
	Qbtu	*Tg C/Qbtu*		*Tg CO_2 eq.*
Diesel	0.377	19.95	1	27.58
Gasoline	0.113	19.33	1	8.02
LP gas	0.080	17.18	1	5.08
Natural gas	0.069	14.47	1	3.64
Electricity	0.156	**	**	27.25
Total	0.795			71.57

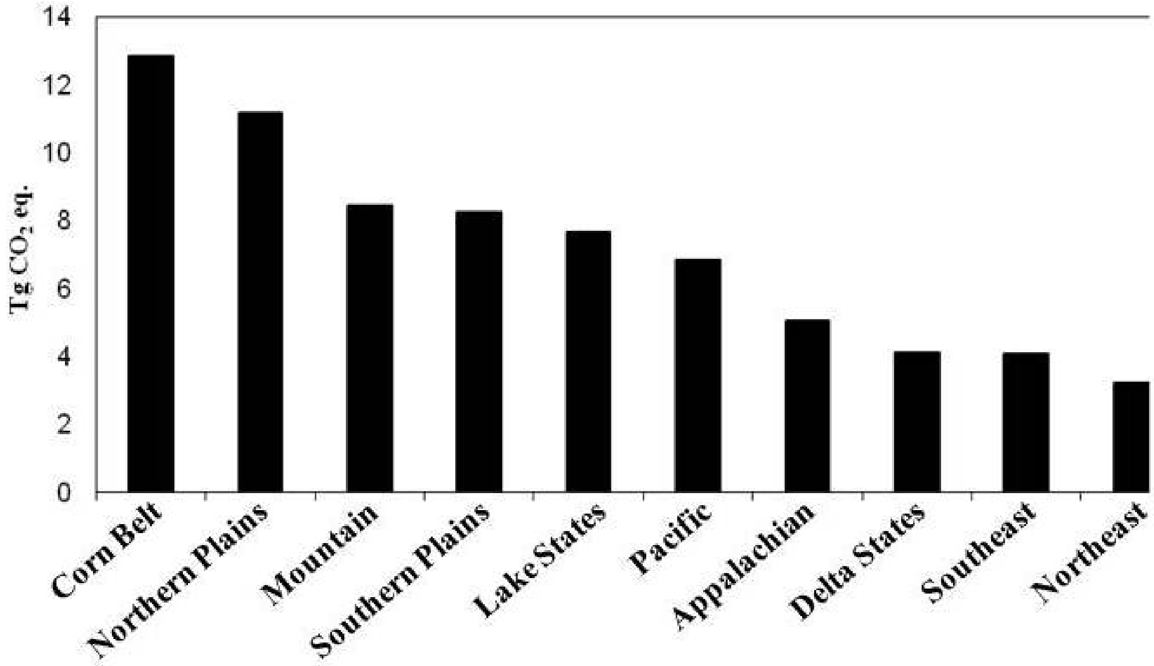
Qbtu is quadrillion British thermal units; Tg C/Qbtu is teragrams carbon per quadrillion British thermal units; Tg CO_2 eq. is teragrams carbon dioxide equivalent; LP is liquefied petroleum; CO_2 is carbon dioxide.
** Varies dependent on fuel used to generate electricity and heat rate of the power generating plant.

5.2 Spatial and Temporal Trends in Greenhouse Gas Emissions from Energy Use in Agriculture

The highest emissions in 2008 were in the Corn Belt and Northern Plains (Figure 5-1). Regions are defined in Table 5-2. Intermediate emissions occurred in the Pacific, Mountain, Southern Plains, and Lake States. Relatively small emissions were estimated for the Southeast, Northeast, Delta, and Appalachian states. There is a strong correlation between production and energy use/emissions. Generally, the states with the most agriculture production use the most energy and therefore have the highest CO_2 emissions (Figure 5-1). However, emissions also vary by the types of energy used for farm production in each region. For example, even though the Pacific region was the third highest energy user among the regions, it ranked sixth in CO_2 emissions (Figure 5-1). The Pacific region has the lowest electricity emission factor among the regions because the western part of the United States has the most hydroelectric power.

Figure 5-1

CO_2 emissions from energy use in agriculture, by region, 2008



Tg CO2 eq. is teragrams carbon dioxide equivalent.

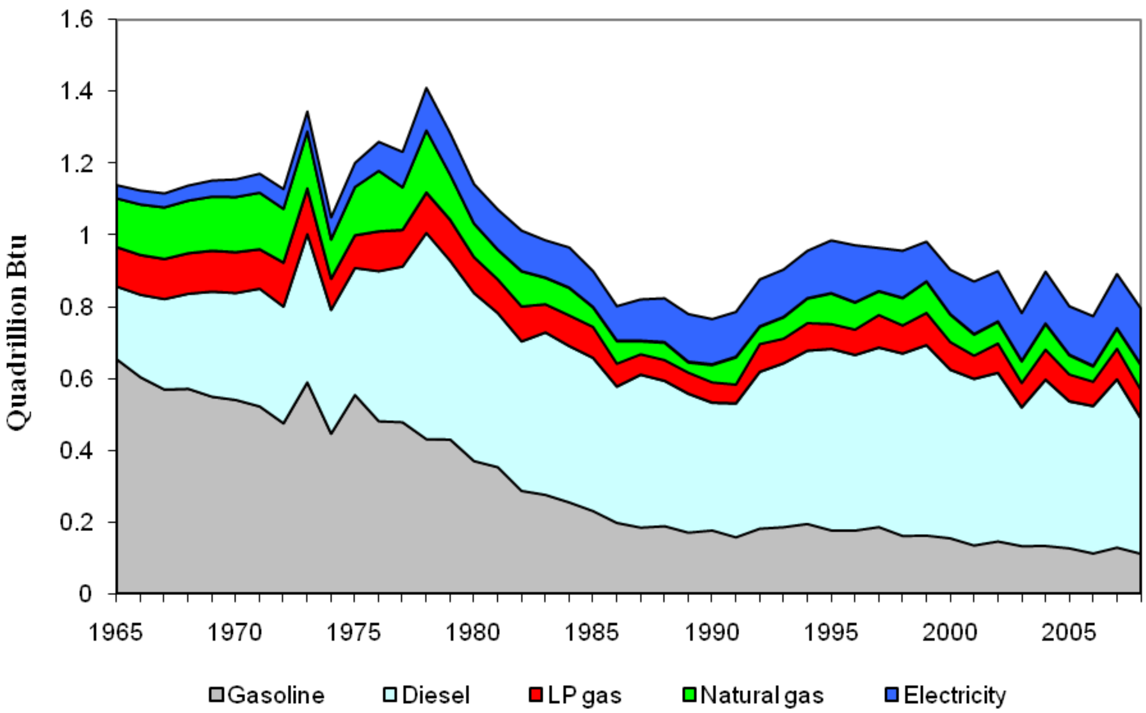
Table 5-2: Definition of Regions Used in Figure 5-1

Region	States of Region	Region	States of Region	Region	States of Region
Corn Belt	Illinois	**Pacific**	California	**Southeast**	Alabama
	Indiana		Oregon		Florida
	Iowa		Washington		Georgia
	Missouri	**Southern Plains**	Oklahoma		South Carolina
	Ohio		Texas	**Northeast**	Connecticut
Mountain	Arizona	**Lake States**	Michigan		Delaware
	Colorado		Minnesota		Maine
	Idaho		Wisconsin		Maryland
	Montana	**Appalachian**	Kentucky		Massachusetts
	Nevada		North Carolina		New Hampshire
	New Mexico		Tennessee		New Jersey
	Utah		Virginia		New York
	Wyoming		West Virginia		Pennsylvania
Northern Plains	Kansas	**Delta States**	Arkansas		Rhode Island
	Nebraska		Louisiana		Vermont
	North Dakota		Mississippi		
	South Dakota				

Agricultural energy use and resulting CO_2 emissions grew throughout the 1960s and 1970s, peaking in the late 1970s (Figure 5-2). High prices, stemming from the oil crisis of the 1970s and early 1980s, drove farmers to be more energy efficient, resulting in a decline in energy use and CO_2 emissions throughout most of the 1980s (Miranowski 2005). This decline is attributed to switching from gasoline-powered to more fuel-efficient diesel-powered engines, adopting energy-conserving tillage practices, shifting to larger multifunction machines, and adopting energy-saving methods of crop drying and irrigation (Uri & Day 1991; Sandretto & Payne 2006; Lin et al. 1995). Another major change in farm energy consumption began around 1979 when automobile manufacturers began to produce more fuel-efficient vehicles. Laws, such as the Energy Policy and Conservation Act of 1975, increased average fuel economy standards, and both gasoline and diesel powered equipment became increasingly energy efficient throughout the 1980s and 1990s. Declines in farm energy use leveled off in the late 1980s as increases in energy prices subsided (Figure 5-2). Energy use increased throughout most of the 1990s, but since 2000, energy use has gone up and down with no apparent trend. However, energy productivity (i.e., output per unit of energy input) has increased significantly.

Figure 5-2

Energy use in agriculture, by source, 1965-2008



Btu is British Thermal Units, or the amount of energy needed to heat 0.454 kg (1 lb) of water from 3.9 °C (39 °F) to 4.4 °C (40 °F).

5.3 Sources of Greenhouse Gas Emissions from Energy Use on Agricultural Operations

Agricultural operations, including crop and livestock farms, dairies, nurseries, and greenhouses, require a variety of energy sources. Energy use in agriculture varies across agricultural operations by crop or livestock type, size of operation, and geographic location. Energy use also varies over time, depending on weather conditions, changes in energy prices, and changes in total annual crop and livestock production. While energy use in agriculture causes CO_2 emissions, this source is small relative to the total U.S. CO_2 emissions from energy.

Different forms of energy are used for different purposes in U.S. agriculture. Energy used on farms is typically categorized as direct and indirect energy (Miranowski 2005). Direct energy is used on the farm for various operations, whereas indirect energy is the energy used to produce energy-intensive farm inputs, such as commercial fertilizers.

Liquid fuel is the most versatile form of direct energy used on farms. Crop production uses large amounts of diesel fuel, gasoline, and liquefied petroleum (LP) gas for field operations. Most large farms use diesel-fueled vehicles for tilling, planting, cultivating, disking, harvesting, and applying fertilizers and pesticides. Gasoline is used for small trucks and older harvesting equipment. Smaller farms are more likely to use gasoline-powered equipment, but as farms get larger they tend to use more diesel fuel.

Farmers use a significant amount of energy to dry crops, such as grain, tobacco, and peanuts. Several types of energy can be used for crop drying, including LP gas, electricity, diesel fuel, and natural gas. Annual rainfall can have a significant effect on the amount of energy used to dry crops from year to year. For example, above-average rainfall, especially just prior to harvest time, can increase the moisture level of grain. In order to meet quality standards, it may require more energy to dry the grain. Weather can also affect the energy used in livestock facilities and other farm buildings that use various forms of energy for heating, cooling, and air circulation. Natural gas is commonly used to control greenhouse temperatures, and dairies rely heavily on electricity to power milking machines and other equipment.

While many irrigation systems in the U.S. are gravity flow systems that require little or no energy for water distribution, irrigation systems that use pumps to distribute water use energy. Based on the 2008 USDA Farm and Ranch Irrigation Survey, about 49 million acres of U.S. farmland were irrigated with pumps powered by liquid fuels, natural gas and electricity, costing a total of $2.68 billion (USDA NASS 2009). Electricity was the principal power source for these pumps, costing $1.5 billion to irrigate about 30 million acres. Diesel fuel was used to power pumps on about 13 million acres and natural gas was used on about 4.7 million acres (USDA NASS 2009).

Irrigated land (including gravity flow irrigation) went up in 2008 to about 55 million acres compared to the 52.6 million acres reported by the 2003 USDA Farm and Ranch Irrigation Survey. Irrigated farmland has been increasing over time, however, and acreage can vary substantially from year to year, depending on environmental conditions and economic factors (Gollehon & Quinby 2006). Corn for grain or seed, soybeans, and alfalfa required the most water in 2008. The leading states in irrigated land in 2008 are Nebraska with 15% of U.S. total, California with 13% of U.S. total, and Texas with almost 10% of U.S. total.

A significant amount of indirect energy is used off the farm to manufacture farm inputs that are ultimately consumed on the farm. Some farm inputs such as fertilizers and pesticides are produced by energy-intensive industries. For example, commercial nitrogen fertilizer is made primarily from natural gas, and synthetic pesticides are made from a variety of chemicals. Although GHG emissions result from the energy consumption used in manufacturing energy-intensive agricultural inputs, these indirect emissions are not detailed in this inventory. For information on the GHG emissions of manufacturing commercial fertilizers see EPA's 2010 U.S. Greenhouse Gas Inventory Report (EPA 2010).

The amount and type of energy used in agricultural operations affect overall CO_2 emissions, and generally CO_2 levels increase with higher energy use in agriculture (Figure 5-2). Some fuels have higher carbon content than others, resulting in higher CO_2 emissions per btu (British thermal unit) used. However, some fuel/engine applications are more energy efficient than others and require less fuel to perform similar operations. For example, diesel fuel has a higher btu content than gasoline on a volumetric basis, but diesel engines have a higher performance rating compared to gasoline engines. Therefore, even though diesel fuel has higher carbon content per btu compared to gasoline, using diesel engines to perform farm operations may result in lower CO_2 emissions.

5.4 Methods for Estimating Carbon Dioxide Emissions from Energy Use in Agriculture

The CO_2 emission estimates for energy use are constructed from fuel consumption data using standardized methods published in the Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990 – 2008 (EPA 2010). Emission estimates from fuel use in agriculture are not explicitly published in the U.S. Greenhouse Gas Inventory Report, however they are contained in the estimates of fuel consumption and emissions by sectors. The emissions estimates presented in this chapter were prepared separately from the U.S. Greenhouse Gas Inventory Report.

Estimates of CO_2 from agricultural operations are based on energy expense data from the Agricultural Resource Management Survey (ARMS) conducted by the National Agricultural Statistics Service (NASS) of the USDA. The ARMS collects information on farm production expenditures, including expenditures on diesel fuel, gasoline, LP gas, natural gas, and electricity (USDA/NASS, 2009). NASS also collects data on price per gallon paid by farmers for gasoline, diesel, and LP gas (USDA/NASS, 2008). Energy expenditures are divided by fuel prices to approximate gallons of fuel consumed by farmers. Gallons of gasoline, diesel, and LP gas are then converted to btu based on the heating value of each of the fuels. The individual farm data are aggregated by state, and the state data are divided into 10 production regions, allowing fuel consumption to be estimated at the national and regional levels. Farm consumption estimates for electricity and natural gas are also approximated by dividing prices into expenditures. Since electricity and natural gas prices are not collected by NASS, we use data from the Energy Information Administration (EIA) that reports average prices by state (EIA, 2009a; EIA 2009b). NASS regional prices were derived by aggregating the EIA state data into NASS production regions.

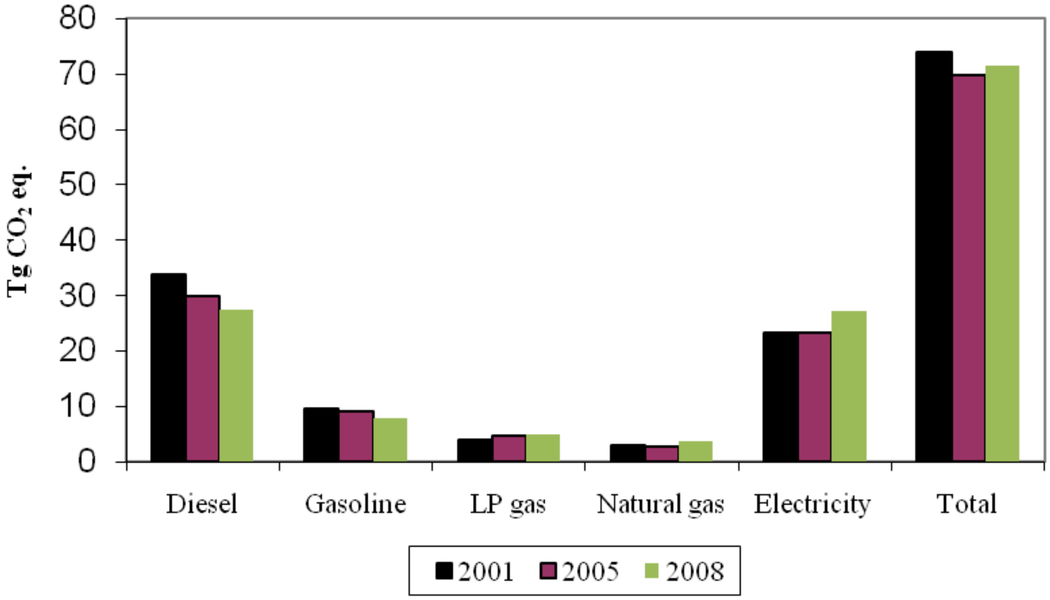
Following the method outlined in Annex 2 of the U.S. Greenhouse Gas Inventory, consumption of diesel fuel, gasoline, LP gas and natural gas were converted to CO_2 emissions using the coefficients for carbon content of fuels and fraction of carbon oxidized during combustion (Table 5-2). These carbon content coefficients were derived by EIA and are similar to those published by the Intergovernmental Panel on Climate Change (IPCC). For each fuel type, fuel consumption in units of quadrillion btu was multiplied by the carbon content coefficient to estimate the Tg of carbon contained in the fuel consumed. This value is sometimes referred to as "potential emissions" because it represents the maximum amount of carbon that could be released to the atmosphere if all carbon were oxidized (EPA 2010). To convert from carbon content to CO_2, it was assumed that the fraction of carbon that is oxidized was 100%.

A different approach was used to estimate emissions from electricity, since a number of fuel sources can be used to generate electricity. Also, fuel sources vary significantly by region, for example, some regions of the country rely more on coal for electricity generation, while other regions use more natural gas to generate electricity. Also, the mix of fuel sources used in a region can change overtime. To account for these variables, the CO_2 emission estimates from electricity generation in this chapter are derived from the most current state data available from EIA. In response to a special request from USDA, EIA tabulated state emission factors for the NASS production regions. The regional-level electricity emission factors represent average CO_2 emissions generated by utility and non-utility electric generators for the 1998-2000 time period. These regional emission factors were multiplied by estimated electricity use in each farm production region to calculate CO_2 emissions. As reported above, electricity use is estimated from farm expenditure data collected by NASS. Price estimates for electricity published by EIA are divided into electricity expenditures to derive the kilowatt hours consumed by farmers. The kilowatt hours of electricity used on the farm are converted to btu, based on a standard conversion rate of 3,413 btu per kilowatt hour.

5.5. Major Changes Compared to Previous Inventories

This report is the third edition of the U.S. Agriculture and Forestry Greenhouse Inventory, which estimates GHG emissions for the year 2008. Figure 5-3 compares the 2008 results with the two previous study periods, 2005 and 2001. Annual GHG emissions are expected to vary with changes in crop and livestock production levels. In addition, weather conditions can have a significant influence on energy use in agriculture, thereby affecting GHG emissions from year to year. Total emissions in 2001 are slightly greater than the previous two reports, with most of the difference related to a higher use of diesel fuel (Figure 5-3). The similarity among the 3 years is an indication that changes in GHG emissions generally follow long-term energy trends as shown in Figure 5-3. When a short-term spike in GHG emissions occurs, it probably is related to a major weather event or other factors significantly affecting agricultural production.

Figure 5-3
CO_2 emissions from energy use in agriculture, by fuel source, 2001, 2005, and 2008



CO_2 is carbon dioxide. LP is liquefied petroleum. Tg CO_2 eq. is teragrams carbon dioxide equivalent.

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Appendix A: Livestock Emissions

Appendix A

Appendix Table A-1 Population of Animals by State in 2008

	Beef Cattle	Dairy Cattle	Swine	Goat	Horse	Sheep	Poultry
State				*Head*			
Alabama	1,288,531	17,984	175,000	50,574	170,731	11,091	209,274,989
Alaska	15,461	799	900	277	5,010	11,091	1,129,194
Arizona	938,433	234,842	165,000	35,374	122,172	155,000	1,129,194
Arkansas	1,822,556	23,978	301,250	32,580	188,414	11,091	241,991,424
California	3,376,306	2,632,471	80,000	103,122	343,973	620,000	45,115,151
Colorado	2,850,938	195,779	750,000	18,561	279,168	420,000	5,016,444
Connecticut	22,921	29,967	3,100	2,586	24,762	8,333	19,283,263
Delaware	11,892	9,191	9,000	1,521	9,001	11,091	45,292,830
Florida	1,663,713	154,889	20,000	39,964	260,451	11,091	24,493,444
Georgia	1,103,618	98,930	230,000	69,498	192,926	11,091	283,713,626
Hawaii	154,364	3,297	13,000	5,364	11,944	11,091	678,444
Idaho	1,443,753	834,036	33,000	11,520	216,945	235,000	1,164,444
Illinois	1,111,217	159,820	4,375,000	17,192	155,495	60,000	6,253,444
Indiana	737,838	242,757	3,600,000	27,801	256,496	52,000	51,345,151
Iowa	4,023,659	324,652	19,525,000	18,898	201,046	225,000	83,990,818
Kansas	7,058,167	184,779	1,800,000	24,763	175,773	90,000	1,129,194
Kentucky	2,270,885	135,858	350,000	68,412	389,629	37,000	62,688,989
Louisiana	891,828	34,975	11,000	14,633	124,575	11,091	18,232,263
Maine	42,550	50,445	4,400	3,162	33,206	8,333	5,316,444
Maryland	125,284	80,924	31,000	9,601	67,647	24,000	57,216,353
Massachusetts	25,058	22,975	10,000	6,022	40,401	8,333	434,444
Michigan	693,922	480,567	1,020,000	21,094	273,584	82,000	27,396,263
Minnesota	1,835,394	732,146	7,525,000	19,768	241,835	145,000	37,610,636
Mississippi	951,906	27,978	375,000	26,738	174,172	11,091	163,385,989
Missouri	4,114,674	169,810	3,112,500	48,654	368,506	82,000	32,169,818
Montana	2,334,925	28,962	175,000	8,613	246,074	270,000	815,444
Nebraska	6,843,288	76,937	3,325,000	11,718	153,615	80,000	13,028,626
Nevada	406,104	36,968	3,500	6,506	42,030	70,000	1,129,194
New Hampshire	16,740	21,479	2,800	3,774	20,662	8,333	663,444
New Jersey	22,817	14,984	8,000	8,312	70,113	11,091	1,942,444
New Mexico	1,152,538	449,621	2,000	19,128	121,702	130,000	1,129,194
New York	531,039	969,909	95,000	33,130	196,216	65,000	21,625,263
North Carolina	816,671	70,927	10,025,000	67,276	167,327	27,000	177,687,788
North Dakota	1,609,165	40,953	151,000	2,523	113,175	93,000	1,129,194
Ohio	1,002,125	383,652	1,897,500	45,061	350,274	125,000	45,704,545
Oklahoma	5,147,362	96,905	2,390,000	82,792	391,394	80,000	48,329,808
Oregon	1,167,345	179,795	20,000	30,628	240,933	220,000	18,824,263
Pennsylvania	874,861	826,131	1,140,000	39,932	295,614	98,000	58,913,878
Rhode Island	3,352	1,698	1,800	468	5,096	8,333	1,129,194
South Carolina	395,427	24,978	240,000	41,192	106,160	11,091	53,460,394
South Dakota	3,378,498	120,889	1,412,500	7,021	181,352	340,000	4,403,667
Tennessee	2,084,215	105,858	205,000	114,664	387,513	30,000	39,377,535
Texas	14,226,582	579,431	1,100,000	1,194,289	970,628	960,000	141,632,535
Utah	700,011	124,874	740,000	9,092	159,976	280,000	5,278,667
Vermont	80,428	197,817	2,600	4,133	29,309	8,333	555,444
Virginia	1,479,084	142,858	350,000	41,275	212,050	81,000	55,693,091
Washington	810,855	347,668	25,000	23,217	197,991	52,000	22,656,263
West Virginia	398,721	15,987	7,000	17,484	83,035	35,000	18,525,485
Wisconsin	1,721,129	1,902,929	385,000	35,179	265,886	90,000	15,713,444
Wyoming	1,240,402	11,984	89,000	5,380	164,011	425,000	318,444
Total	**87,018,553**	**13,658,043**	**67,311,850**	**2,530,466**	**9,499,998**	**5,950,000**	**2,175,119,513**

Source: EPA 2010

Appendix Table A-2 U.S. Livestock Population, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Animal Type						*1 million head*					
Dairy Cattle	**14**	**14**	**13**	**13**	**13**	**13**	**13**	**13**	**13**	**13**	**14**
Dairy Cows	10	9	9	9	9	9	9	9	9	9	9
Dairy Heifers	4	4	4	4	4	4	4	4	4	4	4
Swine	**54**	**59**	**59**	**59**	**60**	**60**	**61**	**61**	**62**	**65**	**67**
Market <60 lbs.	18	20	20	20	20	20	20	20	21	22	22
Market 60-119 lbs.	12	13	13	13	13	13	13	14	14	15	15
Market 120-179 lbs.	9	11	11	11	11	11	11	11	11	12	13
Market >180 lbs.	8	9	9	9	10	10	10	10	10	11	11
Breeding Swine	7	7	6	6	6	6	6	6	6	6	6
Beef cattle	**89**	**97**	**91**	**90**	**89**	**88**	**87**	**87**	**88**	**88**	**87**
Feedlot Steers	6	7	8	8	8	8	8	8	9	9	9
Feedlot Heifers	3	4	5	5	5	5	5	5	5	5	5
Bulls NOF[1]	2	2	2	2	2	2	2	2	2	2	2
Calves NOF	24	25	24	23	23	22	22	22	22	22	22
Heifers NOF	10	12	10	10	10	10	9	10	10	10	9
Steers NOF	10	12	9	9	9	8	8	8	8	8	8
Cows NOF	32	35	34	33	33	33	33	33	33	33	32
Sheep	**11**	**9**	**7**	**7**	**7**	**6**	**6**	**6**	**6**	**6**	**6**
Goats	**3**	**2**	**2**	**2**	**3**	**3**	**3**	**3**	**3**	**3**	**3**
Poultry	**1,537**	**1,827**	**2,033**	**2,060**	**2,098**	**2,085**	**2,131**	**2,150**	**2,154**	**2,167**	**2,175**
Hens >1 yr.	273	299	334	340	340	341	344	348	350	347	340
Pullets	73	81	95	96	95	100	101	97	97	104	99
Chickens	7	8	8	8	8	8	8	8	8	8	8
Broilers	1,066	1,332	1,506	1,525	1,562	1,544	1,589	1,613	1,612	1,619	1,638
Turkeys	118	107	90	91	92	91	88	84	87	89	91
Horses	**5**	**5**	**5**	**6**	**6**	**7**	**8**	**9**	**9**	**9**	**9**

Source: USDA NASS 2009, 2008, 2007, 2006, 2005, 2000, 1995.
Note: Totals may not sum due to independent rounding.
[1](NOF) Not on feed.

Appendix Table A-3 State-Level Methane Emissions from Enteric Fermentation in 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
State						*Tg CO_2 eq.*					
Alabama	1.98	2.21	1.85	1.72	1.74	1.81	1.73	1.69	1.62	1.63	1.57
Alaska	0.01	0.01	0.01	0.01	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Arizona	1.30	1.35	1.33	1.38	1.41	1.37	1.43	1.54	1.61	1.67	1.71
Arkansas	2.19	2.46	2.26	2.23	2.26	2.33	2.41	2.34	2.19	2.22	2.29
California	6.99	7.27	7.98	8.14	8.37	8.02	8.36	8.62	8.69	9.12	9.10
Colorado	3.48	3.96	4.03	4.03	3.98	3.54	3.12	3.39	3.56	3.74	3.79
Connecticut	0.14	0.13	0.12	0.11	0.11	0.09	0.09	0.09	0.09	0.09	0.09
Delaware	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.03	0.04	0.03
Florida	2.82	3.04	2.70	2.65	2.61	2.56	2.60	2.57	2.53	2.61	2.55
Georgia	1.77	1.96	1.71	1.67	1.64	1.70	1.69	1.66	1.63	1.62	1.55
Hawaii	0.27	0.25	0.24	0.22	0.22	0.23	0.23	0.23	0.24	0.24	0.23
Idaho	2.41	2.80	3.11	3.18	3.26	3.22	3.32	3.46	3.55	3.73	3.80
Illinois	2.17	2.13	1.89	1.81	1.78	1.70	1.67	1.72	1.72	1.71	1.63
Indiana	1.56	1.49	1.23	1.16	1.18	1.14	1.15	1.19	1.20	1.26	1.26
Iowa	5.22	5.16	4.76	4.61	4.56	4.62	4.47	4.59	4.82	5.06	5.11
Kansas	6.26	7.56	7.55	7.70	7.77	7.54	7.71	7.73	7.91	7.82	7.97
Kentucky	3.10	3.41	2.85	2.91	2.96	3.03	3.03	2.99	3.05	3.25	3.14
Louisiana	1.44	1.35	1.26	1.20	1.19	1.19	1.22	1.22	1.17	1.21	1.26
Maine	0.19	0.18	0.17	0.16	0.17	0.15	0.15	0.15	0.15	0.15	0.15
Maryland	0.49	0.49	0.41	0.38	0.39	0.36	0.36	0.36	0.36	0.34	0.32
Massachusetts	0.13	0.12	0.10	0.09	0.09	0.08	0.08	0.08	0.08	0.07	0.07
Michigan	1.68	1.70	1.51	1.48	1.48	1.46	1.52	1.54	1.60	1.69	1.71
Minnesota	3.50	3.54	3.30	3.24	3.16	3.01	3.03	3.02	3.00	3.12	3.12
Mississippi	1.67	1.71	1.41	1.42	1.43	1.39	1.36	1.40	1.26	1.24	1.25
Missouri	5.23	5.78	5.31	5.26	5.30	5.47	5.34	5.37	5.60	5.49	5.32
Montana	3.11	3.88	3.78	3.70	3.61	3.57	3.55	3.48	3.55	3.55	3.80
Nebraska	7.11	7.88	8.43	8.38	8.21	8.10	7.96	8.19	8.45	8.76	8.53
Nevada	0.75	0.78	0.77	0.79	0.77	0.78	0.78	0.77	0.77	0.76	0.74
New Hampshire	0.08	0.08	0.08	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
New Jersey	0.11	0.10	0.08	0.08	0.07	0.07	0.07	0.07	0.06	0.06	0.06
New Mexico	1.99	2.36	2.54	2.52	2.54	2.48	2.51	2.57	2.71	2.73	2.84
New York	2.79	2.64	2.70	2.59	2.63	2.48	2.54	2.57	2.59	2.70	2.74
North Carolina	1.20	1.43	1.25	1.26	1.25	1.18	1.17	1.16	1.12	1.13	1.08
North Dakota	2.21	2.69	2.52	2.61	2.61	2.51	2.41	2.41	2.40	2.49	2.42
Ohio	2.04	1.96	1.71	1.71	1.72	1.64	1.70	1.81	1.82	1.80	1.83
Oklahoma	5.71	6.37	5.80	5.68	5.82	6.15	5.81	6.02	6.24	6.15	6.17
Oregon	2.06	2.37	2.23	2.09	2.17	2.11	2.24	2.29	2.25	2.11	2.22
Pennsylvania	2.94	2.80	2.75	2.70	2.68	2.49	2.54	2.58	2.56	2.61	2.62
Rhode Island	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
South Carolina	0.72	0.68	0.61	0.59	0.56	0.56	0.57	0.58	0.55	0.54	0.53
South Dakota	4.21	5.07	4.92	5.14	5.07	4.84	4.82	4.92	4.96	4.92	4.85
Tennessee	2.85	3.17	2.68	2.71	2.76	2.88	2.79	2.76	2.83	2.92	2.77
Texas	16.34	19.44	17.66	17.49	17.58	17.85	17.62	17.60	18.16	18.08	17.95
Utah	1.21	1.37	1.38	1.38	1.38	1.32	1.34	1.34	1.28	1.34	1.39
Vermont	0.58	0.55	0.57	0.56	0.56	0.51	0.53	0.52	0.52	0.52	0.52
Virginia	2.17	2.24	2.05	2.11	2.13	2.08	2.03	2.12	2.17	2.12	2.09
Washington	2.16	2.17	2.09	2.02	1.98	1.86	1.93	1.85	1.91	1.98	1.94
West Virginia	0.60	0.65	0.54	0.52	0.54	0.52	0.50	0.53	0.54	0.56	0.54
Wisconsin	6.00	5.34	5.14	5.06	5.01	4.72	4.96	4.99	5.03	5.25	5.29
Wyoming	1.59	2.00	2.09	2.06	2.00	1.78	1.83	1.80	1.91	1.93	1.82
Total	**126.58**	**138.16**	**131.50**	**130.63**	**130.80**	**128.66**	**128.37**	**130.00**	**132.14**	**134.23**	**133.86**

Source: EPA 2010

Note: State-level emissions do not include data for non-cattle or bulls. Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table A-4 State-Level Methane Emissions from Enteric Fermentation by Livestock Category in 2008

	Beef cattle	Dairy cattle	***Total***
State		*Tg C0₂ eq.*	
Alabama	1.522	0.047	**1.57**
Alaska	0.019	0.002	**0.02**
Arizona	1.053	0.656	**1.71**
Arkansas	2.233	0.052	**2.29**
California	2.390	6.706	**9.10**
Colorado	3.307	0.463	**3.77**
Connecticut	0.015	0.073	**0.09**
Delaware	0.012	0.022	**0.03**
Florida	2.116	0.436	**2.55**
Georgia	1.270	0.284	**1.55**
Hawaii	0.221	0.006	**0.23**
Idaho	1.740	2.048	**3.79**
Illinois	1.280	0.343	**1.62**
Indiana	0.706	0.549	**1.25**
Iowa	4.374	0.728	**5.10**
Kansas	7.519	0.409	**7.93**
Kentucky	2.817	0.325	**3.14**
Louisiana	1.182	0.077	**1.26**
Maine	0.032	0.121	**0.15**
Maryland	0.118	0.201	**0.32**
Massachusetts	0.021	0.053	**0.07**
Michigan	0.533	1.176	**1.71**
Minnesota	1.554	1.570	**3.12**
Mississippi	1.175	0.072	**1.25**
Missouri	4.990	0.331	**5.32**
Montana	3.735	0.060	**3.79**
Nebraska	8.303	0.176	**8.48**
Nevada	0.649	0.093	**0.74**
New Hampshire	0.014	0.055	**0.07**
New Jersey	0.024	0.035	**0.06**
New Mexico	1.638	1.207	**2.85**
New York	0.344	2.399	**2.74**
North Carolina	0.886	0.196	**1.08**
North Dakota	2.334	0.082	**2.42**
Ohio	0.971	0.855	**1.83**
Oklahoma	5.934	0.232	**6.17**
Oregon	1.797	0.417	**2.21**
Pennsylvania	0.571	2.046	**2.62**
Rhode Island	0.004	0.004	**0.01**
South Carolina	0.456	0.069	**0.53**
South Dakota	4.558	0.280	**4.84**
Tennessee	2.510	0.256	**2.77**
Texas	16.404	1.510	**17.91**
Utah	1.088	0.306	**1.39**
Vermont	0.032	0.492	**0.52**
Virginia	1.701	0.385	**2.09**
Washington	1.020	0.912	**1.93**
West Virginia	0.501	0.037	**0.54**
Wisconsin	1.076	4.208	**5.28**
Wyoming	1.793	0.025	**1.82**
Total	**100.54**	**33.09**	**133.63**

Source: EPA 2010. Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table A-5 Cattle Population Categories Used for Estimating Methane Emissions

Dairy Cattle	*Beef Cattle*
Calves	Calves
Heifer Replacements	Heifer Replacements
Cows	Heifer and Steer Stockers
	Animals in Feedlots (Heifers and Steers)
	Cows
	Bulls

Source: EPA 2010

Appendix Table A-6 Dairy Lactation by Region[1]

	California	West	Northern Great Plains	Southcentral	Northeast	Midwest	Southeast
Year				*(kg * year)/cow*			
1990	8,372	66,559	42,812	22,736	77,236	52,303	52,743
1991	8,407	67,689	43,171	22,567	79,184	53,321	53,373
1992	8,492	70,688	44,561	23,321	81,808	54,986	55,075
1993	8,551	70,754	44,780	23,648	81,325	55,167	56,636
1994	9,164	72,957	46,045	24,015	81,694	55,789	57,970
1995	8,878	72,463	46,522	24,001	83,709	57,073	58,719
1996	8,691	73,672	47,248	23,977	84,164	56,593	58,149
1997	8,994	74,496	47,655	23,971	86,676	58,160	59,389
1998	8,823	75,345	49,205	24,167	88,487	59,843	59,252
1999	9,426	75,633	50,450	24,450	89,617	60,676	60,901
2000	9,585	77,056	52,718	25,135	90,412	62,644	62,241
2001	9,482	76,278	52,854	25,002	92,829	61,693	63,078
2002	9,651	78,322	54,951	25,684	94,469	63,499	63,785
2003	9,522	77,600	55,450	26,275	93,256	65,910	62,099
2004	9,589	77,455	55,707	27,711	94,080	66,746	63,946
2005	9,709	78,956	57,794	28,155	95,091	68,757	65,091
2006	9,895	79,414	59,844	27,854	96,716	69,234	65,889
2007	10,179	80,809	60,320	27,459	96,675	69,393	68,011
2008	10,135	80,141	61,700	28,157	98,318	68,914	67,452

Source: USDA 2005d, 2004d, 2003d, 2002d, 2001d, 2000d, 1999a, 1995a.

[1] Beef lactation data developed using methodology described in EPA 2010.

Appendix Table A-7 Typical Livestock Weights

Cattle Type	*lbs.*
Beef Replacement Heifer	
Replacement Weight, 15 Months	715
Replacement Weight, 24 Months	1,078
Mature Weight, 36 Months	1,172
Dairy Replacement Heifer	
Replacement Weight, 15 months	800
Replacement Weight, 24 Months	1,225
Mature Weight, 36 Months	1,350
Stockers– Grazing/Forage Based Only	
Steer Weight Gain/Month to 12 Months	45
Steer Weight Gain/Month to 24 Months	35
Heifer Weight Gain/Month to 12 Months	35
Heifer Weight Gain/Month to 24 Months	30

Source: Feedstuffs (1998), Western Dairyman (1998), Johnson (1999), NRC (1999), EPA 2010.

Appendix Table A-8 U.S. Feedlot Placement in 2008

	Jan	Feb	Mar	Apr	May	Jun	July	Aug	Sep	Oct	Nov	Dec	***Total***
Weight Placed	*Number of animals placed, 1,000 head*												
< 600 lbs.	400	330	335	315	340	365	360	365	445	700	565	425	**4,945**
600 - 700 lbs.	467	385	330	278	350	325	315	410	415	615	630	490	**5,010**
700 - 800 lbs.	525	533	561	428	565	383	481	566	541	543	451	407	**5,984**
> 800 lbs.	395	475	510	515	645	445	500	720	880	580	370	325	**6,360**
Total	**1,787**	**1,723**	**1,736**	**1,536**	**1,900**	**1,518**	**1,656**	**2,061**	**2,281**	**2,438**	**2,016**	**1,647**	**22,299**

Source: USDA (2002f, 2001f, 2000f, 1999a, 1995a), EPA 2010.

Note: Totals may not sum due to independent rounding.

Appendix Table A-9 Regional Estimates of Digestible Energy and Methane Conversion Rates for Enteric Fermentation in 2008

Animal Type	Data	California	West	Northern Great Plains	Southcentral	Northeast	Midwest	Southeast
Beef Repl. Heif.	DE [1]	65	59	66	64	65	65	64
	Ym [2]	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
Dairy Repl. Heif.	DE	64	64	64	64	64	64	64
	Ym	5.6%	5.6%	5.3%	6.0%	5.9%	5.3%	6.5%
Steer Stockers	DE	65	59	66	64	65	65	64
	Ym	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
Heifer Stockers	DE	65	59	66	64	65	65	64
	Ym	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
Steer Feedlot	DE	83	83	83	83	83	83	83
	Ym	3.9%	3.9%	3.9%	3.9%	3.9%	3.9%	3.9%
Heifer Feedlot	DE	83	83	83	83	83	83	83
	Ym	3.9%	3.9%	3.9%	3.9%	3.9%	3.9%	3.9%
Beef Cows	DE	63	57	64	62	63	63	62
	Ym	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
Dairy Cows	DE	67	67	67	67	67	67	67
	Ym	5.5%	5.5%	5.2%	5.9%	5.8%	5.2%	6.4%
Steer Step-Up	DE	72	72	72	72	72	72	72
	Ym	5.2%	5.2%	5.2%	5.2%	5.2%	5.2%	5.2%
Heifer Step-Up	DE	72	72	72	72	72	72	72
	Ym	5.2%	5.2%	5.2%	5.2%	5.2%	5.2%	5.2%

Source: EPA 2010

[1] (DE) Digestible energy; in units of percent gross energy (GE) in MJ/Day.

[2] (Y_m) Methane conversion rate is the fraction of gross energy (GE) in feed converted to methane.

Appendix Table A-10 Definition of Regions in the Enteric Fermentation Model

Region & State(s)

California	Northern Great Plains	Northeast	Southeast	West
California	Colorado	Connecticut	Alabama	Alaska
Midwest	Kansas	Delaware	Florida	Arizona
Illinois	Montana	Maine	Georgia	Hawaii
Indiana	Nebraska	Maryland	Kentucky	Idaho
Iowa	North Dakota	Massachusetts	Mississippi	Nevada
Michigan	South Dakota	New Hampshire	North Carolina	New Mexico
Minnesota	Wyoming	New Jersey	South Carolina	Oregon
Missouri	**South Central**	New York	Tennessee	Utah
Ohio	Arkansas	Pennsylvania	Virginia	Washington
Wisconsin	Louisiana	Rhode Island		
	Oklahoma	Vermont		
	Texas	West Virginia		

Source: EPA 2010

Appendix Table A-11 Methane Emissions from Cattle Enteric Fermentation, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Animal Type						*Gg CH_4*					
Dairy	**1,526**	**1,452**	**1,471**	**1,464**	**1,468**	**1,364**	**1,433**	**1,459**	**1,490**	**1,555**	**1,576**
Cows	1,257	1,197	1,222	1,213	1,218	1,136	1,190	1,210	1,232	1,285	1,302
Replacements 7-11 months	58	55	55	55	55	49	54	56	56	60	60
Replacements 12-23 months	211	199	194	196	195	179	189	194	201	211	214
Beef	**4,502**	**5,128**	**4,790**	**4,757**	**4,761**	**4,762**	**4,680**	**4,731**	**4,803**	**4,837**	**4,799**
Bulls	114	126	122	121	119	119	117	117	120	117	117
Cows	2,887	3,223	3,059	3,041	3,022	3,056	3,037	3,056	3,079	3,083	3,065
Replacements 7-11 months	69	85	74	74	75	76	77	80	82	81	79
Replacements 12-23 months	188	241	204	207	207	214	211	217	228	228	220
Steer Stockers	563	661	508	506	516	484	464	472	475	478	473
Heifer Stockers	305	374	322	320	322	304	291	298	298	294	288
Total Feedlot Cattle	375	416	502	488	499	508	482	489	521	556	557
Total	**6,028**	**6,579**	**6,262**	**6,220**	**6,229**	**6,127**	**6,113**	**6,190**	**6,292**	**6,392**	**6,374**

Source: EPA 2010.

Note: Totals may not sum due to independent rounding. Gg CH_4 is gigagrams methane.

Appendix Table A-12 IPCC Emission Factors for Livestock

	Emission Factors
Animal Type	*(kg CH_4/head/year)*
Dairy	128
Other Cattle	53
Calves	0
Swine	1.5
Sheep	8
Goats	5
Horses	18

Source: EPA 2010, IPCC 2006.
IPCC is the Intergovernmental Panel on Climate Change; kg CH_4 is kilograms methane.

Appendix Table A-13 Summary of Greenhouse Gas Emissions from Managed[1] Waste by State in 2008

	CH_4	N_2O	***Total***
State		*Tg CO_2 eq.*	
Alabama	0.40	0.16	**0.55**
Alaska	0.01	0.00	**0.01**
Arizona	1.00	0.29	**1.29**
Arkansas	0.29	0.21	**0.50**
California	7.18	1.51	**8.68**
Colorado	0.76	0.72	**1.48**
Connecticut	0.03	0.02	**0.06**
Delaware	0.03	0.03	**0.06**
Florida	0.54	0.08	**0.62**
Georgia	0.66	0.23	**0.89**
Hawaii	0.02	0.00	**0.03**
Idaho	1.94	0.57	**2.52**
Illinois	1.19	0.30	**1.49**
Indiana	1.04	0.32	**1.35**
Iowa	6.69	1.41	**8.10**
Kansas	0.94	1.48	**2.42**
Kentucky	0.27	0.10	**0.37**
Louisiana	0.11	0.03	**0.14**
Maine	0.04	0.02	**0.06**
Maryland	0.08	0.07	**0.15**
Massachusetts	0.01	0.01	**0.02**
Michigan	0.84	0.38	**1.22**
Minnesota	2.10	0.78	**2.88**
Mississippi	0.45	0.13	**0.58**
Missouri	0.94	0.24	**1.18**
Montana	0.15	0.06	**0.22**
Nebraska	1.03	1.50	**2.53**
Nevada	0.12	0.03	**0.15**
New Hampshire	0.01	0.01	**0.02**
New Jersey	0.02	0.01	**0.03**
New Mexico	1.31	0.28	**1.59**
New York	0.53	0.31	**0.83**
North Carolina	4.45	0.42	**4.87**
North Dakota	0.11	0.07	**0.18**
Ohio	0.76	0.37	**1.13**
Oklahoma	1.41	0.34	**1.75**
Oregon	0.32	0.15	**0.47**
Pennsylvania	0.60	0.35	**0.95**
Rhode Island	0.00	0.00	**0.01**
South Carolina	0.28	0.06	**0.34**
South Dakota	0.53	0.34	**0.88**
Tennessee	0.20	0.07	**0.27**
Texas	2.65	1.95	**4.61**
Utah	0.46	0.12	**0.58**
Vermont	0.10	0.05	**0.15**
Virginia	0.27	0.12	**0.39**
Washington	0.72	0.26	**0.98**
West Virginia	0.04	0.03	**0.07**
Wisconsin	1.32	1.03	**2.35**
Wyoming	0.07	0.07	**0.14**
Total	**45.02**	**17.11**	**62.13**

Source: EPA 2010. Tg CO_2 eq. is teragrams carbon dioxide equivalent. CH_4 is methane. N_2O is nitrous oxide.
[1]Methane totals include emissions from grazed land manure.

Appendix Table A-14 Methane Emissions from Manure Management by State and Animal in 2008

	Dairy cattle	Beef cattle	Poultry	Swine	Goats	Horses	Sheep	*Total*
State				*Tg CO_2 eq.*				
Alabama	0.0441	0.0148	0.0004	0.0193	0.2546	0.0002	0.0637	**0.3971**
Alaska	0.0003	0.0006	0.0000	0.0004	0.0043	0.0001	0.0001	**0.0058**
Arizona	0.0406	0.8717	0.0003	0.0138	0.0142	0.0025	0.0601	**1.0032**
Arkansas	0.0426	0.0102	0.0003	0.0213	0.1080	0.0002	0.1095	**0.2920**
California	0.0982	6.8946	0.0008	0.0389	0.1061	0.0102	0.0278	**7.1766**
Colorado	0.0767	0.4251	0.0001	0.0211	0.0658	0.0046	0.1637	**0.7571**
Connecticut	0.0004	0.0165	0.0000	0.0019	0.0122	0.0001	0.0003	**0.0314**
Delaware	0.0002	0.0061	0.0000	0.0007	0.0183	0.0001	0.0024	**0.0280**
Florida	0.0599	0.2984	0.0003	0.0295	0.1539	0.0002	0.0024	**0.5445**
Georgia	0.0370	0.1050	0.0005	0.0218	0.4080	0.0002	0.0865	**0.6590**
Hawaii	0.0069	0.0077	0.0000	0.0014	0.0033	0.0002	0.0045	**0.0239**
Idaho	0.0411	1.8665	0.0001	0.0164	0.0127	0.0000	0.0037	**1.9405**
Illinois	0.0274	0.1198	0.0001	0.0117	0.0067	0.0007	1.0216	**1.1880**
Indiana	0.0161	0.1675	0.0001	0.0193	0.0312	0.0006	0.8007	**1.0354**
Iowa	0.1050	0.2629	0.0001	0.0152	0.0465	0.0025	6.2592	**6.6913**
Kansas	0.1856	0.2655	0.0001	0.0133	0.0014	0.0010	0.4693	**0.9362**
Kentucky	0.0541	0.0285	0.0004	0.0294	0.0334	0.0004	0.1279	**0.2740**
Louisiana	0.0331	0.0170	0.0001	0.0141	0.0455	0.0002	0.0021	**0.1121**
Maine	0.0007	0.0232	0.0000	0.0025	0.0088	0.0001	0.0003	**0.0356**
Maryland	0.0026	0.0440	0.0001	0.0051	0.0253	0.0003	0.0068	**0.0842**
Massachusetts	0.0005	0.0087	0.0000	0.0030	0.0008	0.0001	0.0017	**0.0148**
Michigan	0.0142	0.5704	0.0001	0.0206	0.0180	0.0009	0.2134	**0.8376**
Minnesota	0.0378	0.4754	0.0001	0.0182	0.0391	0.0016	1.5277	**2.0999**
Mississippi	0.0339	0.0161	0.0002	0.0197	0.2256	0.0002	0.1533	**0.4490**
Missouri	0.0956	0.0842	0.0003	0.0278	0.0235	0.0009	0.7062	**0.9385**
Montana	0.0649	0.0292	0.0000	0.0186	0.0074	0.0030	0.0318	**0.1548**
Nebraska	0.1921	0.0938	0.0001	0.0116	0.0128	0.0009	0.7177	**1.0290**
Nevada	0.0133	0.1004	0.0000	0.0032	0.0010	0.0008	0.0008	**0.1195**
New Hampshire	0.0003	0.0110	0.0000	0.0016	0.0011	0.0001	0.0003	**0.0144**
New Jersey	0.0005	0.0062	0.0000	0.0053	0.0035	0.0001	0.0015	**0.0172**
New Mexico	0.0360	1.2519	0.0001	0.0092	0.0129	0.0014	0.0001	**1.3115**
New York	0.0089	0.4692	0.0002	0.0148	0.0161	0.0007	0.0169	**0.5269**
North Carolina	0.0177	0.0316	0.0005	0.0189	0.2986	0.0004	4.0787	**4.4465**
North Dakota	0.0420	0.0234	0.0000	0.0085	0.0014	0.0010	0.0305	**0.1068**
Ohio	0.0226	0.2818	0.0002	0.0264	0.0254	0.0014	0.4019	**0.7597**
Oklahoma	0.1211	0.1785	0.0004	0.0295	0.0950	0.0009	0.9873	**1.4128**
Oregon	0.0384	0.2330	0.0002	0.0182	0.0246	0.0024	0.0021	**0.3187**
Pennsylvania	0.0151	0.2724	0.0002	0.0223	0.0316	0.0011	0.2588	**0.6015**
Rhode Island	0.0001	0.0006	0.0000	0.0004	0.0020	0.0001	0.0002	**0.0033**
South Carolina	0.0135	0.0181	0.0003	0.0120	0.1252	0.0002	0.1066	**0.2759**
South Dakota	0.0874	0.1153	0.0000	0.0137	0.0057	0.0037	0.3083	**0.5341**
Tennessee	0.0482	0.0294	0.0006	0.0292	0.0190	0.0003	0.0738	**0.2006**
Texas	0.5353	1.4399	0.0094	0.1098	0.1372	0.0157	0.4053	**2.6527**
Utah	0.0228	0.2497	0.0000	0.0121	0.0587	0.0031	0.1124	**0.4587**
Vermont	0.0010	0.0919	0.0000	0.0022	0.0010	0.0001	0.0001	**0.0963**
Virginia	0.0335	0.0475	0.0002	0.0160	0.0352	0.0009	0.1367	**0.2699**
Washington	0.0250	0.6460	0.0001	0.0149	0.0280	0.0006	0.0034	**0.7180**
West Virginia	0.0096	0.0074	0.0001	0.0063	0.0110	0.0004	0.0010	**0.0357**
Wisconsin	0.0293	1.1895	0.0002	0.0200	0.0101	0.0010	0.0733	**1.3234**
Wyoming	0.0324	0.0113	0.0000	0.0124	0.0007	0.0046	0.0119	**0.0734**
Total	**2.4656**	**19.4292**	**0.0177**	**0.8232**	**2.6325**	**0.0726**	**19.5762**	**45.0170**

Source: EPA 2010. Tg CO_2 eq. is teragrams carbon dioxide equivalent.
Managed manure includes emissions from grazed lands.

Appendix Table A-15 Nitrous Oxide Emissions from Manure Management by State and Animal in 2008

	Dairy cattle	Beef cattle	Poultry	Swine	*Total*
State		*Tg CO_2 eq.*			
Alabama	0.0032	0.0019	0.1395	0.0039	**0.1484**
Alaska	0.0003	0.0000	0.0018	0.0000	**0.0022**
Arizona	0.0937	0.1811	0.0019	0.0035	**0.2802**
Arkansas	0.0032	0.0015	0.1860	0.0075	**0.1981**
California	1.1314	0.2832	0.0512	0.0018	**1.4676**
Colorado	0.0997	0.5705	0.0062	0.0201	**0.6965**
Connecticut	0.0084	0.0001	0.0141	0.0000	**0.0226**
Delaware	0.0024	0.0001	0.0295	0.0002	**0.0322**
Florida	0.0411	0.0012	0.0219	0.0002	**0.0643**
Georgia	0.0189	0.0019	0.1914	0.0055	**0.2177**
Hawaii	0.0014	0.0003	0.0014	0.0003	**0.0034**
Idaho	0.4286	0.1248	0.0019	0.0005	**0.5559**
Illinois	0.0768	0.0900	0.0068	0.1117	**0.2854**
Indiana	0.1023	0.0553	0.0555	0.0887	**0.3019**
Iowa	0.1492	0.6690	0.0834	0.4775	**1.3791**
Kansas	0.0944	1.3219	0.0018	0.0490	**1.4671**
Kentucky	0.0175	0.0081	0.0426	0.0089	**0.0771**
Louisiana	0.0040	0.0010	0.0134	0.0002	**0.0185**
Maine	0.0059	0.0001	0.0059	0.0000	**0.0120**
Maryland	0.0208	0.0047	0.0377	0.0007	**0.0638**
Massachusetts	0.0059	0.0001	0.0011	0.0002	**0.0073**
Michigan	0.2315	0.0860	0.0221	0.0254	**0.3650**
Minnesota	0.3451	0.1594	0.0694	0.1803	**0.7541**
Mississippi	0.0037	0.0018	0.1085	0.0089	**0.1229**
Missouri	0.0702	0.0337	0.0415	0.0743	**0.2198**
Montana	0.0148	0.0216	0.0015	0.0044	**0.0423**
Nebraska	0.0347	1.3581	0.0131	0.0826	**1.4884**
Nevada	0.0170	0.0036	0.0018	0.0001	**0.0224**
New Hampshire	0.0056	0.0000	0.0013	0.0000	**0.0070**
New Jersey	0.0038	0.0001	0.0026	0.0002	**0.0066**
New Mexico	0.1899	0.0778	0.0019	0.0000	**0.2696**
New York	0.2593	0.0142	0.0165	0.0021	**0.2920**
North Carolina	0.0118	0.0016	0.1548	0.2407	**0.4089**
North Dakota	0.0184	0.0342	0.0018	0.0040	**0.0584**
Ohio	0.1569	0.0951	0.0455	0.0442	**0.3417**
Oklahoma	0.0438	0.0204	0.0335	0.0606	**0.1584**
Oregon	0.0743	0.0416	0.0138	0.0003	**0.1300**
Pennsylvania	0.2021	0.0380	0.0559	0.0283	**0.3243**
Rhode Island	0.0004	0.0000	0.0018	0.0000	**0.0022**
South Carolina	0.0038	0.0006	0.0473	0.0065	**0.0582**
South Dakota	0.0554	0.2032	0.0078	0.0359	**0.3024**
Tennessee	0.0161	0.0023	0.0265	0.0050	**0.0500**
Texas	0.2600	1.4839	0.0992	0.0265	**1.8697**
Utah	0.0612	0.0170	0.0087	0.0192	**0.1062**
Vermont	0.0502	0.0002	0.0012	0.0000	**0.0517**
Virginia	0.0213	0.0152	0.0519	0.0088	**0.0972**
Washington	0.1454	0.0860	0.0176	0.0004	**0.2494**
West Virginia	0.0037	0.0030	0.0155	0.0001	**0.0224**
Wisconsin	0.8598	0.1247	0.0128	0.0087	**1.0060**
Wyoming	0.0054	0.0371	0.0010	0.0027	**0.0462**
Total	**5.475**	**7.278**	**1.772**	**1.650**	**16.1746**

Source: EPA 2010.
Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table A-16 Waste Characteristics Data

Livestock	Average TAM[1] (kg)	Source	Nitrogen, N_{ex}[2] (kg/day per 1,000 kg mass)	Source	Max Methane Generation Potential (B_o (m^3 CH4/kg VS added))	Source	Volatile Solids (VS) (kg/day per 1,000 kg mass)	Source
Dairy Cows	604	Safley 2000	0.44	USDA 1996	0.24	Morris 1976	8.8	Lieberman and Pape, 2005
Dairy Heifers	476	Safley 2000	0.31	USDA 1996	0.17	Bryant et. al. 1976	6.7	Lieberman and Pape, 2005
Feedlot Steers	420	USDA 1996	0.3	USDA 1996	0.33	Hashimoto 1981	3.86	Lieberman and Pape, 2005
Feedlot Heifers	420	USDA 1996	0.3	USDA 1996	0.33	Hashimoto 1981	3.98	Lieberman and Pape, 2005
Bulls NOF[3]	750	Safley 2000	0.31	USDA 1996	0.17	Hashimoto 1981	6.04	USDA 1996
Calves NOF	118	ERG 2003	0.3	USDA 1996	0.17	Hashimoto 1981	6.41	USDA 1996
Heifers NOF	420	USDA 1996	0.31	USDA 1996	0.17	Hashimoto 1981	7.09	Lieberman and Pape, 2005
Steers NOF	318	Safley 2000	0.31	USDA 1996	0.17	Hashimoto 1981	7.93	Lieberman and Pape, 2005
Cows NOF	533	NRC 2000	0.33	USDA 1996	0.17	Hashimoto 1981	6.97	Lieberman and Pape, 2005
Market Swine <60 lbs.	16	Safley 2000	0.6	USDA 1996	0.48	Hashimoto 1984	8.8	USDA 1996
Market Swine 60-119 lbs.	41	Safley 2000	0.42	USDA 1996	0.48	Hashimoto 1984	5.4	USDA 1996
Market Swine 120-179 lbs.	68	Safley 2000	0.42	USDA 1996	0.48	Hashimoto 1984	5.4	USDA 1996
Market Swine >180 lbs.	91	Safley 2000	0.42	USDA 1996	0.48	Hashimoto 1984	5.4	USDA 1996
Breeding Swine	198	Safley 2000	0.24	USDA 1996	0.48	Hashimoto 1984	2.6	USDA 1996
Feedlot Sheep	25	EPA 1992	0.42	ASAE 1999	0.36	EPA 1992	9.2	EPA 1992
Sheep NOF	80	EPA 1992	0.42	ASAE 1999	0.19	EPA 1992	9.2	EPA 1992
Goats	64	ASAE 1999	0.45	ASAE 1999	0.17	EPA 1992	9.5	EPA 1992
Horses	450	ASAE 1999	0.3	ASAE 1999	0.33	EPA 1992	10	EPA 1992
Hens ≥ 1 yr	1.8	ASAE 1999	0.83	USDA 1996	0.39	Hill 1982	10.8	USDA 1996
Pullets	1.8	ASAE 1999	0.62	USDA 1996	0.39	Hill 1982	9.7	USDA 1996
Other Chickens	1.8	ASAE 1999	0.83	USDA 1996	0.39	Hill 1982	10.8	USDA 1996
Broilers	0.9	ASAE 1999	1.1	USDA 1996	0.36	Hill 1984	15	USDA 1996
Turkeys	6.8	ASAE 1999	0.74	USDA 1996	0.36	Hill 1984	9.7	USDA 1996

Source: EPA 2010. B_o is maximum methane producing capacity for domestic waste water. CH_4 is methane.
[1](TAM) Typical animal mass.
[2]Nitrogen excretion source.
[3](NOF) Not on feed.

Appendix Table A-17 State Volatile Solids Production Rates in 2008

	Dairy Cow	Dairy Heifer	Beef Cow NOF[1]	Beef Heifer NOF	Beef Steer NOF	Beef Heifer OF[1]	Beef Steer OF
State				*kg/day/1,000 kg mass*			
Alabama	8.40	8.35	7.02	7.81	8.22	4.74	4.27
Alaska	7.30	8.35	9.02	10.05	10.81	4.58	4.15
Arizona	10.37	8.35	9.02	10.34	10.81	4.27	3.91
Arkansas	7.59	8.35	7	7.86	8.19	4.35	3.98
California	10.02	8.35	6.85	7.95	8.00	4.33	3.96
Colorado	10.25	8.35	6.46	7.69	7.52	4.34	3.97
Connecticut	9.22	8.35	6.9	7.67	8.07	4.93	4.41
Delaware	8.63	8.35	6.9	7.72	8.07	4.64	4.19
Florida	8.90	8.35	7.02	7.75	8.22	4.58	4.15
Georgia	9.07	8.35	7.02	7.85	8.22	4.63	4.18
Hawaii	7.00	8.35	9.02	10.26	10.81	4.58	4.15
Idaho	10.11	8.35	9.02	10.82	10.81	4.42	4.03
Illinois	9.07	8.35	6.91	8.07	8.07	4.59	4.15
Indiana	9.38	8.35	6.91	7.98	8.07	4.35	3.98
Iowa	9.46	8.35	3.91	8.27	8.07	4.28	3.93
Kansas	9.63	8.35	6.46	7.75	7.52	4.35	3.97
Kentucky	7.89	8.35	7.02	7.91	8.22	4.65	4.20
Louisiana	7.39	8.35	7	7.73	8.19	4.48	4.07
Maine	8.99	8.35	6.9	7.76	8.07	4.47	4.07
Maryland	9.02	8.35	6.9	7.76	8.07	4.45	4.05
Massachusetts	8.63	8.35	6.9	7.74	8.07	4.58	4.15
Michigan	10.05	8.35	6.91	7.99	8.07	4.38	4.00
Minnesota	9.17	8.35	6.91	8.04	8.07	4.24	3.89
Mississippi	8.19	8.35	7.02	7.82	8.22	4.57	4.14
Missouri	8.02	8.35	6.91	7.85	8.07	4.49	4.08
Montana	9.03	8.35	6.46	7.17	7.52	4.69	4.23
Nebraska	9.09	8.35	6.46	7.71	7.52	4.35	3.98
Nevada	9.65	8.35	9.02	10.49	10.81	4.48	4.07
New Hampshire	9.44	8.35	6.9	7.74	8.07	4.30	3.94
New Jersey	8.51	8.35	6.9	7.89	8.07	4.36	3.98
New Mexico	10.34	8.35	9.02	10.56	10.81	4.22	3.88
New York	9.42	8.35	6.9	8.02	8.07	4.05	3.75
North Carolina	9.38	8.35	7.02	7.83	8.22	4.65	4.20
North Dakota	8.40	8.35	6.46	7.43	7.52	4.22	3.88
Ohio	9.01	8.35	6.91	7.93	8.07	4.33	3.96
Oklahoma	8.58	8.35	7	8.08	8.19	4.35	3.98
Oregon	9.40	8.35	9.02	10.54	10.81	4.46	4.06
Pennsylvania	9.26	8.35	6.9	8.00	8.07	4.35	3.98
Rhode Island	8.94	8.35	6.9	7.60	8.07	4.87	4.36
South Carolina	9.05	8.35	7.02	7.81	8.22	4.58	4.15
South Dakota	9.45	8.35	6.46	7.50	7.52	4.39	4.01
Tennessee	8.60	8.35	7.02	7.86	8.22	5.02	4.48
Texas	9.51	8.35	7	8.21	8.19	4.32	3.95
Utah	9.70	8.35	9.02	10.51	10.81	4.22	3.88
Vermont	9.03	8.35	6.9	7.89	8.07	4.52	4.10
Virginia	9.02	8.35	7.02	7.87	8.22	4.35	3.98
Washington	10.36	8.35	9.02	10.77	10.81	4.47	4.07
West Virginia	8.13	8.35	6.9	7.74	8.07	5.25	4.65
Wisconsin	9.34	8.35	6.91	7.87	8.07	4.31	3.95
Wyoming	9.29	8.35	6.46	7.30	7.52	4.61	4.17

Source: EPA 2010.

[1](NOF) Not on feed or (OF) On feed.

Appendix Table A-18 State-Based Methane Conversion Factors[1] for Liquid Waste Management Systems in 2008

State	Liquid/Slurry and Deep Pit	Anaerobic Lagoon
	percent	
Alabama	0.38	0.75
Alaska	0.14	0.46
Arizona	0.43	0.76
Arkansas	0.35	0.74
California	0.36	0.75
Colorado	0.22	0.65
Connecticut	0.24	0.68
Delaware	0.31	0.73
Florida	0.51	0.76
Georgia	0.37	0.75
Hawaii	0.58	0.76
Idaho	0.22	0.65
Illinois	0.28	0.71
Indiana	0.27	0.70
Iowa	0.25	0.68
Kansas	0.30	0.73
Kentucky	0.31	0.73
Louisiana	0.46	0.76
Maine	0.20	0.62
Maryland	0.30	0.72
Massachusetts	0.24	0.67
Michigan	0.24	0.67
Minnesota	0.23	0.67
Mississippi	0.24	0.75
Missouri	0.30	0.72
Montana	0.20	0.61
Nebraska	0.26	0.70
Nevada	0.26	0.71
New Hampshire	0.21	0.64
New Jersey	0.29	0.71
New Mexico	0.28	0.70
New York	0.23	0.65
North Carolina	0.34	0.73
North Dakota	0.21	0.64
Ohio	0.27	0.70
Oklahoma	0.36	0.74
Oregon	0.21	0.64
Pennsylvania	0.26	0.69
Rhode Island	0.26	0.69
South Carolina	0.39	0.75
South Dakota	0.24	0.68
Tennessee	0.33	0.73
Texas	0.43	0.76
Utah	0.22	0.66
Vermont	0.21	0.63
Virginia	0.30	0.72
Washington	0.21	0.64
West Virginia	0.26	0.69
Wisconsin	0.23	0.66
Wyoming	0.20	0.62

Source: EPA 2010, IPCC 2006.

[1](MCF) Methane conversion factors represent weighted average of multiple animal types.

Appendix Table A-19 Additional Methane and Nitrous Oxide Emission Factors

	Methane	Nitrous Oxide
Manure Management System		
Pasture	0.015	0.02
Daily spread	0.005	0
Solid storage	0.04	0.005
Dry lot	0.015	0.02
Poultry with bedding	0.015	0.001
Poultry without bedding	0.015	0.001
Liquid/Slurry	0.25	0.005
Aerobic treatments	0	0.005

Source: IPCC 2006.

Appendix Table A-20 State-Weighted Methane Conversion Factors for Livestock Waste Emissions in 2008[1]

State	Beef Feed Lot Heifer	Beef Feed Lot Steer	Dairy Cow	Dairy Heifer	Swine Market	Swine Breeding	Layer	Broiler	Turkey	Sheep	Goats	Horses
						percent						
Alabama	0.01	2.09	0.69	0.02	2.49	0.54	8.69	3.41	0.03	0.01	0.02	0.92
Alaska	0.00	0.02	0.03	0.00	0.00	0.00	0.18	+	0.03	0.01	0.00	0.02
Arizona	0.75	1.19	41.37	0.14	2.37	0.49	0.65	+	0.03	0.12	0.01	0.66
Arkansas	0.01	2.02	0.47	0.02	2.39	2.83	0.54	3.72	0.89	0.01	0.01	1.01
California	1.39	3.29	325.90	2.41	1.03	0.29	4.32	0.27	0.46	0.48	0.04	1.85
Colorado	1.51	2.14	20.12	0.12	5.15	2.65	3.11	+	0.03	0.22	0.00	1.00
Connecticut	0.00	0.02	0.77	0.02	0.01	0.01	0.28	0.27	0.03	0.00	0.00	0.09
Delaware	0.00	0.01	0.29	0.01	0.07	0.05	0.07	0.78	0.03	0.01	0.00	0.03
Florida	0.01	2.84	14.10	0.12	0.06	0.05	7.10	0.20	0.03	0.01	0.02	1.40
Georgia	0.01	1.75	4.93	0.07	2.99	1.13	14.89	4.52	0.03	0.01	0.03	1.04
Hawaii	0.00	0.33	0.36	0.00	0.14	0.08	0.13	+	0.03	0.01	0.00	0.06
Idaho	0.33	1.63	88.35	0.53	0.12	0.06	0.58	+	0.03	0.12	0.00	0.78
Illinois	0.25	1.05	5.70	0.11	39.37	9.28	0.29	+	0.03	0.03	0.00	0.56
Indiana	0.16	0.61	7.83	0.14	32.66	5.47	0.80	0.27	0.41	0.03	0.01	0.92
Iowa	1.87	3.13	12.32	0.20	269.44	28.62	1.68	0.27	0.26	0.12	0.00	0.72
Kansas	3.79	5.05	12.51	0.13	18.95	3.39	0.04	+	0.03	0.05	0.01	0.63
Kentucky	0.03	2.55	1.26	0.09	4.96	1.13	0.59	0.98	0.03	0.02	0.02	1.40
Louisiana	0.01	1.57	0.78	0.03	0.07	0.03	1.86	0.27	0.03	0.01	0.01	0.67
Maine	0.00	0.03	1.07	0.03	0.01	0.01	0.39	+	0.03	0.00	0.00	0.12
Maryland	0.01	0.11	2.05	0.05	0.25	0.07	0.23	0.95	0.03	0.01	0.00	0.24
Massachusetts	0.00	0.02	0.40	0.02	0.06	0.02	0.01	+	0.03	0.00	0.00	0.15
Michigan	0.24	0.44	26.91	0.25	8.39	1.77	0.56	0.27	0.03	0.04	0.01	0.98
Minnesota	0.44	1.36	22.15	0.49	62.43	10.32	0.35	0.14	1.37	0.08	0.00	0.87
Mississippi	0.01	1.60	0.75	0.02	6.06	1.24	8.02	2.69	0.03	0.01	0.01	0.94
Missouri	0.10	4.46	3.90	0.11	26.17	7.46	0.25	0.27	0.60	0.04	0.01	1.32
Montana	0.06	3.03	1.37	0.02	1.19	0.32	0.33	+	0.03	0.14	0.00	0.88
Nebraska	3.79	5.36	4.43	0.04	27.49	6.69	0.56	0.02	0.03	0.04	0.00	0.55
Nevada	0.01	0.62	4.77	0.02	0.03	0.01	0.02	+	0.03	0.04	0.00	0.15
New Hampshire	0.00	0.01	0.51	0.01	0.01	0.00	0.03	+	0.03	0.00	0.00	0.07
New Jersey	0.00	0.02	0.28	0.01	0.06	0.02	0.14	+	0.03	0.01	0.00	0.25
New Mexico	0.21	1.50	59.40	0.21	0.00	0.00	0.59	+	0.03	0.07	0.00	0.44
New York	0.04	0.38	21.68	0.66	0.62	0.18	0.47	0.27	0.03	0.03	0.01	0.70
North Carolina	0.01	0.84	1.46	0.05	159.74	34.48	10.52	2.55	1.15	0.02	0.03	0.90
North Dakota	0.09	1.91	1.09	0.03	0.85	0.60	0.04	+	0.03	0.05	0.00	0.41
Ohio	0.27	0.81	13.22	0.20	16.11	3.03	0.85	0.18	0.17	0.07	0.01	1.26
Oklahoma	0.49	5.27	8.42	0.08	33.71	13.30	3.74	0.76	0.03	0.04	0.02	1.41
Oregon	0.13	1.70	10.97	0.13	0.07	0.03	0.87	0.27	0.03	0.11	0.01	0.87
Pennsylvania	0.12	0.60	12.43	0.54	10.61	1.72	0.66	0.51	0.33	0.05	0.01	1.06
Rhode Island	+	0.00	0.03	0.00	0.01	0.00	0.07	+	0.03	0.00	0.00	0.02
South Carolina	0.00	0.64	0.84	0.02	4.50	0.57	4.84	0.76	0.36	0.01	0.02	0.57
South Dakota	0.56	3.60	5.43	0.06	11.93	2.75	0.14	+	0.13	0.18	0.00	0.65
Tennessee	0.01	2.29	1.31	0.09	2.98	0.54	0.24	0.64	0.03	0.02	0.03	1.39
Texas	5.95	19.54	68.09	0.48	16.29	3.01	4.45	2.05	0.03	0.75	0.45	5.23
Utah	0.05	1.04	11.82	0.07	4.07	1.28	2.68	+	0.12	0.15	0.00	0.57
Vermont	0.00	0.05	4.27	0.11	0.00	0.00	0.02	+	0.03	0.00	0.00	0.11
Virginia	0.05	1.55	2.17	0.09	5.59	0.92	0.36	0.80	0.52	0.04	0.01	0.76
Washington	0.27	0.92	30.56	0.21	0.12	0.04	1.04	0.27	0.03	0.03	0.01	0.71
West Virginia	0.01	0.45	0.35	0.01	0.03	0.01	0.14	0.27	0.11	0.02	0.00	0.30
Wisconsin	0.34	1.05	55.46	1.18	2.80	0.69	0.29	0.17	0.03	0.05	0.01	0.95
Wyoming	0.10	1.45	0.53	0.01	0.29	0.28	0.01	+	0.03	0.22	0.00	0.59

Source: EPA 2010

[1](MCFs) Methane conversion factors are weighted by the distribution of waste management systems for each animal type within a state.

Appendix Table A-21 Nitrogen in Livestock Waste on Grazed Lands

Year	*Teragrams Nitrogen*
1990	3.9
1991	3.9
1992	4.0
1993	4.0
1994	4.0
1995	4.1
1996	4.0
1997	3.9
1998	3.8
1999	3.8
2000	3.7
2001	3.7
2002	3.7
2003	3.7
2004	3.7
2005	3.8
2006	3.8
2007	3.8
2008	3.7

Source: EPA 2010

Appendix B: Crop Emissions

Appendix B

Appendix Table B-1 Rice Harvested Area, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
State and Crop						*1,000 Hectares*					
Arkansas	**486**	**542**	**571**	**656**	**608**	**589**	**629**	**662**	**567**	**536**	**565**
Primary	486	542	571	656	608	589	629	662	567	536	565
Ratoon	0	0	0	0	0	0	0	1	0	0	0
California	**160**	**188**	**222**	**191**	**214**	**205**	**239**	**213**	**212**	**216**	**209**
Florida	**7**	**15**	**11**	**7**	**8**	**5**	**7**	**5**	**6**	**8**	**7**
Primary	5	10	8	5	5	2	4	5	5	6	5
Ratoon	2	5	3	3	3	2	3	0	1	2	2
Louisiana	**287**	**276**	**329**	**281**	**283**	**294**	**280**	**264**	**279**	**303**	**269**
Primary	221	206	251	214	251	231	216	236	251	249	194
Ratoon	66	69	78	66	32	64	65	28	28	54	75
Mississippi	**101**	**117**	**88**	**102**	**102**	**95**	**95**	**106**	**76**	**76**	**93**
Missouri	**32**	**45**	**68**	**84**	**74**	**69**	**79**	**87**	**87**	**72**	**81**
Texas	**200**	**180**	**130**	**122**	**114**	**101**	**119**	**103**	**84**	**80**	**106**
Primary	143	129	87	87	83	73	88	81	61	59	70
Ratoon	57	51	43	35	31	28	31	22	24	21	37
Total	**1,273**	**1,363**	**1,418**	**1,443**	**1,403**	**1,358**	**1,448**	**1,440**	**1,310**	**1,291**	**1,330**

Appendix Table B-2 Total U.S. Production of Crops Managed with Burning, 1990, 1995, 2000-2008

	1990	1995	2000	2001	2002	2003	2004	2005	2006	2007	2008
Crop						*Million Metric Tons*					
Wheat	74.3	59.4	60.6	53.0	43.7	63.8	58.7	57.3	49.3	55.8	68.0
Rice	7.1	7.9	8.7	9.8	9.6	9.1	10.6	10.2	8.8	9.0	9.3
Sugarcane	25.5	27.9	32.8	31.4	32.3	30.7	26.3	24.1	26.8	27.2	27.8
Corn	201.5	188.0	251.9	241.4	227.8	256.3	299.9	282.3	267.6	331.2	307.4
Barley	9.2	7.8	6.9	5.4	4.9	6.1	6.1	4.6	3.9	4.6	5.2
Soybeans	52.4	59.2	75.1	78.7	75.0	66.8	85.0	83.4	86.8	72.9	80.5
Peanuts	1.6	1.6	1.5	1.9	1.5	1.9	1.9	2.2	1.6	1.7	2.3
Total	**371.7**	**351.8**	**437.5**	**422.0**	**394.8**	**434.6**	**488.6**	**464.1**	**444.8**	**502.3**	**500.6**

Appendix Table B-3 Production of Crops Managed with Burning

	Corn	Soybeans	Barley	Wheat	Peanuts	Rice	Sugarcane
Year		*1,000 bushels*			*1,000 lbs.*	*1,000 cwt*	*1,000 tons*
1990	7,933,894	2,063,480	361,876	2,924,713	3,603,593	156,830	25,525
1991	7,474,639	2,128,399	397,987	2,121,542	4,926,477	160,554	27,444
1992	9,476,538	2,346,768	390,071	2,642,953	4,284,347	180,876	27,545
1993	6,337,623	2,003,236	341,172	2,567,571	3,392,350	157,379	28,188
1994	10,050,350	2,694,457	321,305	2,486,723	4,247,377	199,104	28,057
1995	7,399,926	2,329,519	308,031	2,338,576	3,461,420	175,197	27,922
1996	9,232,401	2,550,250	336,365	2,440,017	3,661,133	172,767	26,729
1997	9,206,676	2,880,755	308,462	2,658,669	3,539,330	184,000	28,766
1998	9,758,520	2,936,751	301,340	2,729,227	3,963,382	185,505	31,486
1999	9,430,452	2,843,264	233,135	2,459,487	3,829,431	207,052	32,023
2000	9,914,883	2,954,747	272,399	2,387,274	3,265,454	191,903	32,762
2001	9,502,419	3,097,107	212,850	2,086,522	4,276,631	215,925	31,377
2002	8,966,635	2,952,965	194,488	1,720,555	3,320,988	211,672	32,253
2003	10,089,051	2,628,882	238,524	2,512,200	4,144,070	200,265	30,715
2004	11,806,886	3,346,750	239,775	2,312,366	4,288,117	232,920	26,320
2005	11,113,894	3,281,984	181,622	2,254,987	4,869,775	223,769	24,137
2006	10,534,690	3,415,921	154,425	1,941,434	3,464,198	194,292	26,820
2007	13,037,654	2,868,291	180,091	2,197,557	3,672,178	199,148	27,187
2008	12,101,033	3,170,490	205,280	2,678,016	5,147,817	204,409	27,842

Appendix Table B-4 Information Used in Estimating Methane and Nitrous Oxide Emissions from Crop Residue Burning in 2008

B-4(a) Crop Assumptions and Coefficients

Assumption/Coefficient	Corn	Peanuts	Soybeans	Barley	Wheat	Rice	Sugarcane
Residue/Crop Ratio	1	1	2.1	1.2	1.3	1.4	0.19
Fraction Residue Burned	0.03	0.03	0.03	0.03	0.03	variable	0.95
Fraction Dry Matter	0.91	0.86	0.87	0.93	0.93	0.91	0.62
Burning Efficiency	0.93	0.93	0.93	0.93	0.93	0.93	0.93
Combustion Efficiency	0.88	0.88	0.88	0.88	0.88	0.88	0.88
Fraction Carbon	0.4478	0.45	0.45	0.4485	0.4428	0.3806	0.4235
Fraction Nitrogen	0.0058	0.0106	0.023	0.0077	0.0062	0.0072	0.004

B-4(b) Emissions Factors and Global Warming Potentials

Greenhouse Gas	Factor & Global Warming Potential
Emissions Factor	
Methane	0.005
Nitrous Oxide	0.007
Global Warming Potential	
Methane	21
Nitrous Oxide	310

B-4(c) Rice Area Burned by State

State	% Burned
Arkansas	20
California	11
Florida[1]	0
Louisiana	5
Mississippi	25
Missouri	3
Texas	0

[1]Crop residue burning is illegal in Florida.

Appendix Table B-5 Soil Carbon Stocks by Climate Region and U.S. Soil Groupings[1]

IPCC	USDA	CTD	CTM	WTD	WTM	STD	STM
Inventory Soil Categories	**Taxonomic Soil Orders**	*Metric Tons C/ha*					
High Clay Activity Mineral Soils	Vertisols, Mollisols, Inceptisols, Aridisols, & High Base Status Alfisols	42	65	37	51	42	57
Low Clay Activity Mineral Soils	Ultisols, Oxisols, Acidic Alfisols, & Many Entisols	45	52	25	40	39	47
Sandy Soils	>70% Sand, <8% Clay	24	40	16	30	33	50
Volcanic Soils	Andisols	124	114	124	124	124	128
Spodic Soils	Spodosols	86	74	86	107	86	86
Aquic Soils	Soils With Aquic Suborder	86	89	48	51	63	48
Organic Soils	Histosols[2]	n/a	n/a	n/a	n/a	n/a	n/a

IPCC is Intergovernmental Panel on Climate Change. C/ha is Carbon per hectare.
[1]U.S. soil groupings are based on the IPCC Soil Inventory categories and the USDA taxonomic soil orders.

[2]Carbon stocks are not needed for organic soils and are thus represented by n/a or "not applicable".
Note: Carbon stocks are for the top 30 cm of the soil profile, and were estimated from pedon data available in the NSSC database (NRCS 1997).
Climate regions: Cold temperate dry (CTD), cold temperate moist (CTM), warm temperate dry (WTD), warm temperate moist (WTM), subtropical temperate dry (STD), and subtropical temperate moist (STM).

Appendix Table B-6 Stock Change Factors for the U.S. and IPCC Default Values for Impacts on Mineral Soils

Factors	IPCC Default	Warm Moist Climate	Warm Dry Climate	Cool Moist Climate	Cool Dry Climate
Land Use Change					
Cultivated[1]	1	1	1	1	1
General Uncultivated[1]	1.4	1.42 ± 0.06	1.37 ± 0.05	1.24 ± 0.06	1.20 ± 0.06
Set Aside	1.25	1.31 ± 0.06	1.26 ± 0.04	1.14 ± 0.06	1.10 ± 0.05
Improved Grassland					
Medium Input	1.1	1.14 ± 0.06	1.14 ± 0.06	1.14 ± 0.06	1.14 ± 0.06
High Input	n/a[2]	1.11 ± 0.04	1.11 ± 0.04	1.11 ± 0.04	1.11 ± 0.04
Wetland Rice Production	1.1	1.1	1.1	1.1	1.1
Tillage					
Conventional Till	1	1	1	1	1
Reduced Till	1.05	1.08 ± 0.03	1.09 ± 0.09	1.08 ± 0.03	1.01 ± 0.03
No-till	1.1	1.13 ± 0.02	1.17 ± 0.08	1.13 ± 0.02	1.05 ± 0.03
Cropland Input					
Low	0.9	0.94 ± 0.01	0.94 ± 0.01	0.94 ± 0.01	0.94 ± 0.01
Medium	1	1	1	1	1
High (without manure)	1.1	1.07 ± 0.02	1.07 ± 0.02	1.07 ± 0.02	1.07 ± 0.02
High (with manure)	1.2	1.38 + 0.06	1.34 ± 0.08	1.38 ± 0.06	1.34 ± 0.08

IPCC is Intergovernmental Panel on Climate Change.
[1]Factors in the IPCC report (2006) were converted to represent changes in soil organic content storage from a cultivated condition rather than a native condition.

[2]n/a indicates "non applicable".

Appendix Table B-7 Cultivated Histosol (Organic Soils) Area

	Temperate	Sub-Tropical
Year		*1,000 ha*
1990	444	194
1991	444	194
1992	444	194
1993	450	196
1994	450	196
1995	450	196
1996	450	196
1997	450	196
1998	450	196
1999	450	196
2000	450	196
2001	450	196
2002	450	196
2003	450	196
2004	450	196
2005	450	196
2006	450	196
2007	450	196
2008	450	196

Appendix Table B-8 Carbon Loss Rates from Organic Soils Under Agricultural Management in the United States

	Cropland	Grassland[1]
Climate Regions	*Metric Tons C/ha-yr*	
CTD & CTM	11.2 ± 2.5	2.8 ± 0.51
WTD & WTM	14.0 ± 2.5	3.5 ± 0.81
STD & STM	14.0 ± 3.3	3.5 ± 0.81

C/ha-yr is carbon per hectare per year.

[1]There is not enough data available to estimate values for C losses from grasslands. Estimates are 25% of the values for cropland (the IPCC default organic soil C losses on grasslands).

Climate regions: Cold temperate dry (CTD), cold temperate moist (CTM), warm temperate dry (WTD), warm temperate moist (WTM), subtropical temperate dry (STD), and subtropical temperate moist (STM).

Appendix Table B-9 State-Level Estimates of Annual Soil Carbon Stock Changes by Major Land Use and Management Type, 2008

	Net Change, Cropland [1]	Net Change, Hay	CRP	Ag. Land on Organic Soils	***Total*** [2]
State			*Tg CO_2 eq.*		
Alabama	(0.14)	(0.15)	(0.12)	0.00	**(0.42)**
Alaska	ND	ND	ND	ND	**0.00**
Arizona	0.04	(0.01)	0.00	0.00	**0.03**
Arkansas	0.43	(0.10)	(0.06)	0.00	**0.28**
California	(0.14)	0.19	(0.02)	2.27	**2.30**
Colorado	0.33	(0.25)	(0.13)	0.00	**(0.04)**
Connecticut	(0.02)	(0.01)	0.00	0.00	**(0.03)**
Delaware	0.02	0.00	(0.00)	0.00	**0.02**
Florida	0.55	0.10	(0.01)	10.00	**10.64**
Georgia	0.14	(0.10)	(0.02)	0.00	**0.03**
Hawaii	ND	ND	ND	0.29	**0.29**
Idaho	0.29	(0.34)	(0.52)	0.09	**(0.49)**
Illinois	(2.71)	(0.36)	(0.77)	0.50	**(3.34)**
Indiana	(0.53)	(0.30)	(0.18)	2.80	**1.78**
Iowa	(1.39)	(0.57)	(1.48)	0.73	**(2.71)**
Kansas	(0.98)	(0.64)	(1.00)	0.00	**(2.61)**
Kentucky	0.12	(0.47)	(0.12)	0.00	**(0.47)**
Louisiana	(0.57)	(0.08)	(0.09)	0.00	**(0.74)**
Maine	0.03	(0.09)	(0.03)	0.00	**(0.09)**
Maryland	0.03	(0.10)	(0.00)	0.03	**(0.04)**
Massachusetts	(0.00)	(0.02)	0.00	0.03	**0.01**
Michigan	(0.33)	(0.45)	(0.12)	2.29	**1.39**
Minnesota	0.45	(0.73)	(0.81)	5.72	**4.64**
Mississippi	0.10	(0.11)	(0.28)	0.00	**(0.29)**
Missouri	(0.47)	(1.66)	(1.50)	0.00	**(3.63)**
Montana	(1.69)	(1.59)	(2.09)	0.03	**(5.34)**
Nebraska	(0.50)	(0.37)	(0.51)	0.00	**(1.39)**
Nevada	(0.02)	(0.01)	0.00	0.00	**(0.03)**
New Hampshire	0.00	(0.03)	0.00	0.01	**(0.02)**
New Jersey	0.01	(0.09)	0.00	0.01	**(0.07)**
New Mexico	0.12	(0.08)	(0.14)	0.00	**(0.11)**
New York	0.43	(1.03)	(0.06)	0.54	**(0.13)**
North Carolina	(0.04)	(0.12)	(0.03)	2.22	**2.04**
North Dakota	(2.48)	(1.81)	(1.73)	0.00	**(6.03)**
Ohio	(1.05)	(0.75)	(0.26)	0.41	**(1.65)**
Oklahoma	(0.55)	(0.34)	(0.22)	0.00	**(1.10)**
Oregon	0.23	(0.19)	(0.27)	0.06	**(0.16)**
Pennsylvania	0.04	(1.03)	(0.06)	0.00	**(1.04)**
Rhode Island	0.00	0.00	0.00	0.00	**0.00**
South Carolina	0.08	(0.07)	(0.03)	0.04	**0.02**
South Dakota	(1.15)	(1.41)	(0.84)	0.00	**(3.40)**
Tennessee	(0.19)	(0.45)	(0.14)	0.00	**(0.79)**
Texas	(0.10)	0.04	(0.75)	0.00	**(0.80)**
Utah	(0.16)	0.17	(0.09)	0.00	**(0.08)**
Vermont	0.08	(0.15)	0.00	0.00	**(0.06)**
Virginia	0.04	(0.45)	(0.01)	0.02	**(0.40)**
Washington	0.26	(0.50)	(0.67)	0.08	**(0.82)**
West Virginia	(0.03)	(0.27)	0.00	0.00	**(0.30)**
Wisconsin	0.17	(0.45)	(0.57)	2.15	**1.30**
Wyoming	0.06	(0.19)	(0.04)	0.01	**(0.15)**
Total	**(11.16)**	**(17.41)**	**(15.78)**	**30.35**	**(14.00)**

[1] Annual cropping systems on mineral soils (e.g., corn, soybean, cotton, and wheat).

[2] Total does not include change in soil organic carbon storage on federal lands, including those that were previously under private ownership, and does not include carbon storage due to sewage sludge applications.

Tg CO_2 eq. is teragrams carbon dioxide equivalent. ND= No data.

Appendix Table B-10 State-Level Estimates of Mineral Soil Carbon Changes on Cropland[1] by Major Activity, 2008

	Cropland Remaining Cropland		Grassland Converted to Cropland		
	Irrigated	Non-irrigated	Irrigated	Non-irrigated	***Net Total***
State			*Tg CO_2 eq.*		
Alabama	(0.01)	(0.54)	0.00	0.14	**(0.40)**
Alaska	ND	ND	ND	ND	**0.00**
Arizona	0.02	(0.00)	0.00	0.00	**0.02**
Arkansas	0.21	(0.06)	0.06	0.10	**0.30**
California	0.09	(0.10)	0.12	(0.04)	**0.07**
Colorado	0.06	(0.09)	0.00	(0.01)	**(0.04)**
Connecticut	0.00	(0.03)	0.00	0.00	**(0.03)**
Delaware	(0.01)	0.01	0.00	0.00	**0.01**
Florida	(0.01)	(0.00)	0.14	0.51	**0.64**
Georgia	(0.03)	(0.09)	0.00	0.17	**0.06**
Hawaii	ND	ND	ND	ND	**0.00**
Idaho	0.14	(0.73)	0.08	(0.06)	**(0.58)**
Illinois	(0.11)	(3.76)	0.00	0.06	**(3.81)**
Indiana	(0.03)	(1.01)	(0.00)	0.08	**(0.96)**
Iowa	(0.03)	(2.95)	(0.01)	(0.40)	**(3.38)**
Kansas	0.31	(2.79)	(0.01)	(0.09)	**(2.59)**
Kentucky	(0.02)	(0.44)	0.00	0.05	**(0.40)**
Louisiana	(0.23)	(0.55)	0.01	0.05	**(0.72)**
Maine	0.01	(0.11)	0.00	0.00	**(0.10)**
Maryland	(0.01)	(0.07)	0.00	0.02	**(0.05)**
Massachusetts	(0.00)	(0.02)	0.00	0.00	**(0.03)**
Michigan	(0.00)	(0.64)	0.00	(0.09)	**(0.73)**
Minnesota	0.03	(0.84)	(0.01)	(0.13)	**(0.95)**
Mississippi	0.05	(0.57)	0.00	0.23	**(0.29)**
Missouri	(0.02)	(2.81)	0.04	(0.76)	**(3.55)**
Montana	(0.44)	(4.40)	(0.09)	(0.36)	**(5.29)**
Nebraska	0.43	(1.82)	0.08	(0.07)	**(1.39)**
Nevada	(0.03)	(0.01)	0.01	0.00	**(0.03)**
New Hampshire	0.00	(0.02)	0.00	0.00	**(0.02)**
New Jersey	0.01	(0.10)	0.00	0.00	**(0.08)**
New Mexico	(0.04)	(0.06)	0.03	(0.04)	**(0.11)**
New York	(0.01)	(0.70)	0.00	0.04	**(0.66)**
North Carolina	0.01	(0.25)	0.00	0.00	**(0.23)**
North Dakota	(0.08)	(5.76)	(0.00)	(0.17)	**(6.02)**
Ohio	(0.01)	(1.72)	0.00	(0.25)	**(1.98)**
Oklahoma	(0.01)	(1.01)	0.00	0.03	**(0.99)**
Oregon	0.10	(0.29)	0.04	(0.06)	**(0.21)**
Pennsylvania	0.01	(0.88)	0.00	(0.13)	**(1.00)**
Rhode Island	ND	ND	ND	ND	**0.00**
South Carolina	(0.01)	(0.09)	0.00	0.09	**(0.01)**
South Dakota	0.01	(3.08)	(0.01)	(0.29)	**(3.37)**
Tennessee	(0.03)	(0.87)	0.00	0.14	**(0.75)**
Texas	(0.19)	(0.64)	0.03	0.04	**(0.75)**
Utah	(0.06)	(0.07)	0.05	0.00	**(0.07)**
Vermont	0.00	(0.04)	0.00	(0.00)	**(0.05)**
Virginia	0.01	(0.35)	0.00	(0.07)	**(0.41)**
Washington	(0.08)	(0.81)	0.06	(0.07)	**(0.90)**
West Virginia	0.00	(0.21)	0.00	(0.08)	**(0.28)**
Wisconsin	(0.01)	(0.63)	(0.00)	(0.11)	**(0.76)**
Wyoming	(0.03)	(0.06)	0.01	(0.07)	**(0.15)**
Total	**(0.00)**	**(42.03)**	**0.64**	**(1.60)**	**(42.99)**

1 Data from mineral soils used; includes soil C sequestration on CRP lands.

[2]Losses from annual cropping systems due to plow-out of pastures, rangeland, hayland, and perennial/horticultural cropland.

Tg CO_2 eq. is teragrams carbon dioxide equivalent. ND= No data.

Note: Parenthesis indicate a net sequestration.

Appendix C: Forest Carbon Stocks

Appendix C

Appendix Table C-1 Forest Area, Stock, and Stock Change by State[1]

State	Forest Area	Net Area Change	Non-Soil Stocks	SOC[2]	Non-Soil Change	Harvested Wood Products Change
	1000 ha	*1000 ha yr^{-1}*	*Tg CO_2 eq.*	*Tg CO_2 eq.*	*Tg CO_2 eq. yr^{-1}*	
Alabama	9,184	(28.3)	2,455	1445	(4.4)	(6.4)
Alaska	6,192	n/a	4,510	1970	n/a	(0.4)
Arizona	7,558	(24.7)	1,601	620	7.4	(0.3)
Arkansas	7,396	(27.0)	2,301	1138	(3.5)	(4.2)
California	13,511	54.2	7,989	1908	(66.4)	(3.7)
Colorado	9,244	48.2	3,373	1035	(20.5)	(0.2)
Connecticut	707	(6.7)	333	159	1.6	(0.1)
Delaware	148	(1.1)	65	34	(0.2)	(0.1)
Florida	6,838	26.1	1,727	2580	(22.4)	(3.1)
Georgia	10,046	3.8	2,942	3020	(28.2)	(7.1)
Idaho	8,622	(11.8)	3,854	1294	7.5	(1.5)
Illinois	1,831	13.3	723	356	(4.1)	(0.6)
Indiana	1,920	14.3	826	372	(14.1)	(0.6)
Iowa	1,227	34.8	415	253	(10.8)	(0.2)
Kansas	902	15.6	273	268	(5.6)	(0.1)
Kentucky	4,860	(17.9)	1,745	719	(3.8)	(1.2)
Louisiana	5,722	11.5	1,639	965	0.9	(4.4)
Maine	7,161	(1.4)	2,698	2157	(8.3)	(3.3)
Maryland	987	(8.2)	481	225	(5.2)	(0.4)
Massachusetts	1,236	(3.8)	620	315	(2.6)	(0.3)
Michigan	7,815	1.0	2,930	4176	6.8	(2.2)
Minnesota	6,876	64.1	1,964	4201	(16.2)	(1.9)
Mississippi	7,941	31.8	2,236	1270	(21.5)	(5.5)
Missouri	6,231	73.2	2,217	1098	(40.6)	(1.0)
Montana	10,360	67.5	4,161	1497	(16.2)	(1.3)
Nebraska	504	13.5	157	129	(3.4)	(0.1)
Nevada	4,488	21.0	900	354	(2.6)	(0.1)
New Hampshire	1,914	(4.9)	912	518	(0.4)	(0.2)
New Jersey	843	(2.7)	327	207	(1.5)	(0.2)
New Mexico	6,753	21.9	1,774	574	(14.2)	(0.1)
New York	7,645	20.7	3,418	1977	(23.1)	(1.4)
North Carolina	7,525	25.6	2,655	1878	(25.2)	(5.0)
North Dakota	289	(1.2)	72	81	1.1	(0.0)
Ohio	3,205	2.0	1,253	742	(11.6)	(0.4)
Oklahoma	3,102	37.2	829	465	(15.7)	(0.8)
Oregon	12,176	8.3	7,276	3543	(28.1)	(6.2)
Pennsylvania	6,707	(10.6)	2,867	1546	(13.6)	(1.3)
Rhode Island	148	(1.8)	65	35	(0.4)	(0.0)
South Carolina	5,218	47.5	1,589	1532	(20.8)	(3.5)
South Dakota	757	23.4	191	156	(4.6)	(0.2)
Tennessee	5,659	26.3	2,113	838	(21.2)	(2.0)
Texas[2]	24,363	(0.1)	3,521	4451	(0.1)	(4.0)
Utah	7,374	94.2	1,873	702	(13.1)	(0.1)
Vermont	1,850	(2.3)	881	499	2.5	(0.4)
Virginia	6,364	(9.7)	2,473	1318	(12.4)	(3.1)
Washington	9,061	22.7	6,248	2881	(28.8)	(5.6)
West Virginia	4,857	(0.4)	2,172	1059	(19.6)	(1.0)
Wisconsin	6,757	70.1	2,274	3486	(24.9)	(2.4)
Wyoming	4,633	10.7	1,715	501	(5.8)	(0.1)
Total	**276,706**	**740**	**101,632**	**62,544**	**(558)**	**(88)**

[1] Net change values are model outputs for 2008 (Smith et al. 2010); stocks and area are based on the most recent inventory per state. Parentheses indicate negative values, which are a net decrease of forest area or net increase in carbon sequestration.
A value of "n/a indicates not available. Tg CO_2 eq. is teragrams carbon dioxide equivalent.
[2] SOC (soil organic carbon) does not include effects of past land use history.
[3] The central and western portions of these states are represented by a single survey; therefore estimates of change are based only on forests in the eastern forests in the eastern portions of the states.
portions of the states.

Appendix Table C-2 Carbon Stock Pools on Private Forestland by Region and Age-Class[1]

	Age Class	SOC[2]	Dead Plant Matter	Biomass	*Total*
Region	*Years*		*Tg CO_2 eq.*		
North		**16,399**	**5,159**	**15,523**	**37,080**
	<20	1,420	273	275	**1,967**
	20-40	2,408	528	1,409	**4,346**
	40-60	4,648	1,359	4,332	**10,339**
	60-80	4,892	1,770	5,695	**12,356**
	80-100	2,232	902	2,808	**5,942**
	100-150	740	305	953	**1,998**
	150-200	37	14	32	**83**
	200+	3	1	3	**8**
	Unknown	19	7	17	**42**
South		**18,620**	**4,650**	**19,006**	**42,276**
	<20	5,968	1,093	2,774	**9,835**
	20-40	4,656	1,059	4,116	**9,831**
	40-60	4,080	1,196	5,475	**10,751**
	60-80	2,450	818	4,247	**7,516**
	80-100	814	274	1,403	**2,490**
	100-150	267	85	410	**762**
	150-200	16	5	22	**43**
	Unknown	371	119	558	**1,047**
Pacific Coast		**3,587**	**2,444**	**4,536**	**10,567**
	<20	775	353	217	**1,345**
	20-40	654	360	740	**1,754**
	40-60	660	447	1,042	**2,149**
	60-80	586	435	891	**1,912**
	80-100	383	314	628	**1,325**
	100-150	291	266	533	**1,091**
	150-200	67	71	147	**285**
	200+	88	87	151	**326**
	Unknown	83	111	187	**381**
Rocky Mountain		**1,487**	**1,647**	**1,981**	**5,116**
	<20	322	330	140	**792**
	20-40	79	74	56	**209**
	40-60	127	122	136	**385**
	60-80	236	241	321	**797**
	80-100	267	305	445	**1,016**
	100-150	299	376	587	**1,262**
	150-200	99	129	190	**418**
	200+	58	71	107	**236**
Total		**40,093**	**13,899**	**41,047**	**95,039**

[1] Stocks are based on the most recent inventory per state.
[2] SOC (soil organic carbon) does not include effects of past land use history.
Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table C-3 Carbon Stock Pools on Public Forestland by Region and Age-Class[1]

Region	Age class	SOC[2]	Dead Plant Matter	Biomass	***Total***
	Years		*Tg CO_2 eq.*		
North		**7,649**	**1,977**	**5,475**	**15,101**
	<20	762	100	108	**969**
	20-40	951	146	361	**1,459**
	40-60	1,625	344	972	**2,941**
	60-80	2,246	670	1,972	**4,889**
	80-100	1,339	472	1,431	**3,243**
	100-150	649	216	559	**1,424**
	150-200	61	20	49	**130**
	200+	8	4	13	**25**
	Unknown	8	3	10	**21**
South		**2,998**	**807**	**3,762**	**7,567**
	<20	437	64	159	**660**
	20-40	531	102	386	**1,020**
	40-60	695	189	874	**1,758**
	60-80	818	271	1,365	**2,454**
	80-100	356	121	660	**1,138**
	100-150	121	45	252	**418**
	150-200	1	1	4	**6**
	Unknown	38	14	61	**113**
Pacific Coast		**6,714**	**6,084**	**12,958**	**25,757**
	<20	443	255	118	**816**
	20-40	569	302	503	**1,374**
	40-60	516	324	654	**1,494**
	60-80	785	601	1,289	**2,674**
	80-100	744	630	1,365	**2,740**
	100-150	1,171	1,133	2,573	**4,877**
	150-200	722	790	1,778	**3,289**
	200+	1,714	1,987	4,577	**8,278**
	Unknown	52	62	101	**215**
Rocky Mountain		**5,090**	**6,360**	**9,263**	**20,712**
	<20	825	814	314	**1,953**
	20-40	216	205	158	**579**
	40-60	198	197	235	**630**
	60-80	548	595	923	**2,065**
	80-100	832	1,003	1,676	**3,512**
	100-150	1,416	1,975	3,310	**6,701**
	150-200	708	1,060	1,791	**3,558**
	200+	346	511	857	**1,714**
Total		**22,452**	**15,228**	**31,458**	**69,138**

[1] Stocks are based on the most recent inventory per state.

[2] SOC (soil organic carbon) does not include effects of past land use history.

Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table C-4 Carbon Stock Pools on Timberlands by Region and Stand Size Class[1]

	Stand Size Class	SOC[2]	Dead Plant Matter	Biomass	***Total***
Region			*Tg CO_2 eq.*		
North		**22,786**	**6,725**	**19,930**	**49,441**
	Nonstocked	286	40	12	**338**
	Seedling/ Sapling	4,768	843	1,045	**6,656**
	Poletimber	7,523	2,011	5,190	**14,723**
	Sawtimber	10,209	3,831	13,684	**27,723**
South		**17,573**	**4,753**	**20,812**	**43,138**
	Nonstocked	273	29	23	**325**
	Seedling/ Sapling	4,042	727	1,438	**6,207**
	Poletimber	4,606	1,158	4,676	**10,441**
	Sawtimber	8,652	2,838	14,674	**26,165**
Pacific Coast		**7,401**	**5,756**	**12,370**	**25,527**
	Nonstocked	180	113	26	**319**
	Seedling/ Sapling	1,128	539	320	**1,987**
	Poletimber	825	451	809	**2,086**
	Sawtimber	5,267	4,653	11,215	**21,135**
Rocky Mountain		**3,631**	**4,534**	**7,182**	**15,346**
	Nonstocked	181	175	42	**398**
	Seedling/ Sapling	527	483	303	**1,313**
	Poletimber	729	752	1,205	**2,686**
	Sawtimber	2,194	3,123	5,632	**10,949**
Total		**51,391**	**21,767**	**60,294**	**133,452**

[1] Stocks are based on the most recent inventory per state.

[2] SOC (soil organic carbon) does not include effects of past land use history.

Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table C-5 Carbon Stocks[1] on all Forestland by Forest Type Group and Ownership Class[2]

	Private	Public	Reserve/Other
Forest Type Group		*Tg CO_2 eq.*	
East	**42,264**	**9,956**	**4,138**
Aspen/Birch	1,026	623	96
Elm/Ash/Cottonwood	2,036	384	184
Loblolly/Shortleaf Pine	5,335	823	46
Longleaf/Slash Pine	860	284	12
Maple/Beech/Birch	6,977	1,910	698
Oak/Gum/Cypress	2,934	654	158
Oak/Hickory	17,239	3,282	1,506
Oak/Pine	2,876	603	100
Pinyon/Juniper	8	3	424
Spruce/Fir	1,338	688	172
White/Red/Jack Pine	1,131	498	109
Woodland Hardwoods	2	0	460
Other Hardwood Type Groups	109	43	24
Other Softwood Type Groups	309	141	39
Nonstocked	84	20	109
West	**8,447**	**21,395**	**15,433**
Alder/Maple	437	198	30
Aspen/Birch	265	797	229
California Mixed Conifer	528	1,566	546
Douglas-fir	2,894	6,557	1,266
Fir/Spruce/Mountain Hemlock	683	4,927	3,860
Hemlock/Sitka Spruce	768	2,491	1,567
Lodgepole Pine	163	1,428	801
Other Western Softwoods	61	313	663
Pinyon/Juniper	1	8	3,602
Ponderosa Pine	1,067	1,825	244
Redwood	240	15	119
Spruce/Fir	13	9	72
Tanoak/Laurel	428	216	126
Western Larch	50	243	31
Western Oak	488	414	887
Western White Pine	2	15	31
Woodland Hardwoods	14	22	911
Other Hardwood Type Groups	214	122	114
Nonstocked	129	227	333
Total	**50,710**	**31,351**	**19,571**

[1] Excluding soils.

[2] SOC (soil organic carbon) Stocks are based on the most recent inventory per state.

Note: Other Hardwood Type Groups and Other Softwood Type Groups represent aggregates of minor type groups. However, "Other Western Softwoods" is a specific type group within the Forest Inventory Analysis DataBase (FIADB).

Note: The Private and Public ownership classes represent timberlands only. The Reserved or Other (lower productivity) forests include both public and private owners.

Tg CO_2 eq. is teragrams carbon dioxide equivalent.

Appendix Table C-6 Net Annual Carbon Stock Change[1] on all Forestland by Forest Type Group and Ownership Class[2]

	Private	Public	Reserve/Other
Forest Type Group		*Tg CO_2 eq. yr^{-1}*	
East			
Aspen/Birch	6.3	6.1	1.2
Elm/Ash/Cottonwood	3.8	(4.9)	1.6
Loblolly/Shortleaf Pine	(87.0)	(17.7)	2.6
Longleaf/Slash Pine	(1.7)	(8.2)	(0.7)
Maple/Beech/Birch	(3.3)	(32.2)	4.3
Oak/Gum/Cypress	32.2	(23.5)	0.4
Oak/Hickory	(209.5)	(83.4)	17.3
Oak/Pine	47.2	(4.2)	1.9
Pinyon/Juniper	n/a	n/a	n/a
Spruce/Fir	8.8	(10.0)	2.0
White/Red/Jack Pine	13.8	(8.0)	2.6
Woodland Hardwoods	n/a	n/a	n/a
Other Hardwood Type Groups	(5.6)	(1.7)	(2.1)
Other Softwood Type Groups	(9.1)	(0.6)	0.7
Nonstocked	n/a	n/a	n/a
West			
Alder/Maple	n/a	n/a	n/a
Aspen/Birch	n/a	n/a	n/a
California Mixed Conifer	n/a	n/a	n/a
Douglas-fir	n/a	n/a	n/a
Fir/Spruce/Mountain Hemlock	n/a	n/a	n/a
Hemlock/Sitka Spruce	n/a	n/a	n/a
Lodgepole Pine	n/a	n/a	n/a
Other Western Softwoods	n/a	n/a	n/a
Pinyon/Juniper	n/a	n/a	(6.9)
Ponderosa Pine	n/a	n/a	n/a
Redwood	n/a	n/a	n/a
Spruce/Fir	n/a	n/a	n/a
Tanoak/Laurel	n/a	n/a	n/a
Western Larch	0.4	(5.6)	0.3
Western Oak	n/a	n/a	n/a
Western White Pine	n/a	n/a	n/a
Woodland Hardwoods	n/a	n/a	(11.5)
Other Hardwood Type Groups	n/a	n/a	n/a
Nonstocked	n/a	n/a	n/a
***Total*[3]**			

[1] Excluding soils.

[2] Net change values are model estimates for 2008 (Smith et al. 2010). Parentheses indicate negative values, which are a net decrease of forest area or net increase in carbon sequestration. A value of "n/a" indicates not available, and totals from columns that include "n/a" should be interpreted accordingly.

[3] Totals would not be the sum of the change of individual forest types because at a more aggregate resolution, more data are available.

Note: Other Hardwood Type Groups and Other Softwood Type Groups represent aggregates of minor type groups. However, "Other Western Softwoods" is a specific type group within the Forest Inventory Analysis DataBase (FIADB).

Note: The Private and Public ownership classes represent timberlands only. The Reserved or Other (lower productivity) forests include both public and private owners.

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