U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2005. Global Change Program Office, Office of the Chief Economist, U.S. Department of Agriculture. Technical Bulletin No. 1921. 161 pp. August, 2008. http://www.usda.gov/oce/global_change/AFGGInventory1990_2005.htm.

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 35%, 155%, and 18%, respectively, since 1750. In the U.S., agriculture accounted for close to 7% of total GHG emissions (7260 Tg CO_2 eq.) in 2005. Livestock, poultry, and crop production contributed a total of 481 Tg CO_2 eq. to the atmosphere in 2005. This total includes an offset from agricultural soil carbon sequestration of roughly 32 Tg CO_2 eq. The primary agricultural sources are N_2O emissions from cropped and grazed soils (263 Tg CO_2 eq.), CH_4 emissions from enteric fermentation (112 Tg CO_2 eq.), and CH_4 emissions from managed livestock waste (41 Tg CO_2 eq.). Forests in the United States contributed a net reduction in atmospheric GHG of approximately 787 Tg CO_2 eq. in 2005, which offset total U.S. GHG emissions by approximately 11%. In aggregate, the U.S. agricultural sector (including GHG sources for crop, poultry, and livestock production and GHG removal from the atmosphere via sinks for in) was estimated to be a net sink of 306 Tg CO_2 eq. in 2005.

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

August 2008

Dear Reader:

I am pleased to present you with this report, *The U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2005,* an update to USDA Technical Bulletin 1907 (2004) which accounted for greenhouse gas emissions and sinks for the agricultural and forestry sectors through 2001.

This report is consistent with the Environmental Protection Agency's (EPA's) *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (2007) 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 USDA's Agricultural Research Service. I also express my thanks to Linda Heath and James Smith of the USDA Forest Service, James Duffield of USDA's Office of Energy Policy and New Uses, 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 Global Change Program Office

Contributors

Stephen Del Grosso (editor), Agricultural Research Service, USDA
Margaret Walsh (co-editor), Global Change Program Office, USDA
James Duffield, Office of Energy Policy and New Uses, USDA
Linda Heath, Forest Service, USDA
Stephen Ogle, Natural Resources Ecology Laboratory, Colorado State University
James Smith, Forest Service, USDA
Tom Wirth, Office of Atmospheric Programs, EPA

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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 and Sadie Skiles of USDA ARS NPA Soil Plant Nutrient 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. 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 their analysis for this report to partition regional and land ownership trends in forest carbon. James Duffield of the USDA 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 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.

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
DM	Dry matter
Btu	British thermal unit
Qbtu	Quadrillion British thermal units
Tbtu	Trillion British thermal units
EF	Emission factor
MCF	Methane conversion factor

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Chapter 1: Introduction

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

Global concentrations of the three most important long-lived greenhouse gases (GHG) in the atmosphere have increased measurably over the past 255 years. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) concentrations in the atmosphere have increased by approximately 35%, 155%, and 18%, respectively, since 1750 (Keeling & Whorf 2005, Dlugokencky et al. 2005, Prinn et al. 2000). Agriculture and forestry practices may either contribute to or remove GHG from the atmosphere. Agriculture and forestry have affected 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. This report serves to estimate U.S. GHG emissions for the agricultural sector, to quantify uncertainty in emission estimates, and to estimate the potential of agriculture to mitigate U.S. GHG emissions.

Observed increases in atmospheric GHG concentrations are primarily a result of fossil fuel combustion for power generation, transportation, and construction. In the U.S., agriculture accounted for close to 7% of total GHG emissions (7,260 Tg CO_2 eq., teragrams of carbon dioxide equivalents) in 2005 (EPA 2007). 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 481 Tg CO_2 eq. to the atmosphere in 2005 (Table 1-1). This total includes an offset, or sink, from agricultural (cropped and grazed lands) soil carbon sequestration of roughly 32 Tg CO_2 eq. Forests in the United States contributed a net reduction in atmospheric GHGs of approximately 787 Tg CO_2 eq. in 2005, which offset total U.S. GHG emissions by almost 11% (EPA 2007). After accounting for C sequestration related to forestry, agricultural and forested lands in the U.S. were estimated to be a net sink of 306 Tg CO_2 eq. (Table 1-1). The 95% confidence interval for this estimate ranges from a sink of 499 to 110 Tg CO_2 eq. (Table 1-1).

Table 1-1 Agriculture and Forestry Greenhouse Gas Emission Estimates and Uncertainty Intervals, 2005

	Estimate	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Source		*Tg CO_2 eq.*			%
Livestock	162	148	184	(9)	14
Crops[1]	153	137	188	(11)	23
Grassland	96	79	143	18	48
Energy Use[2]	69				
Forestry	(699)	(890)	(513)	(27)	27
Urban Trees	(89)				
Net Emissions	**(306)**	**(499)**	**(110)**	**(63)**	**64**

Note: Parentheses indicate net sequestration.

[1] Includes sequestration in agricultural soils.

[2] Confidence intervals were not available for this component.

A little more than one-third (35%) of agriculture's GHG emissions in 2005 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 22% from enteric fermentation,

BOX 1-1

The USDA GHG Inventory report follows the international convention for reporting greenhouse gas emissions, as described in the introduction of the U.S. GHG Inventory (EPA 2006). Emissions of greenhouse gases are expressed in equivalent terms, normalized to carbon dioxide using Global Warming Potentials (GWPs) published by the IPCC (IPCC SAR). Global Warming Potentials, which are based on physical and chemical properties of gases, represent the relative effect of a given greenhouse gas 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 greenhouse gas 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 values and estimated atmospheric lifetime were revised for some gases in the IPCC Third Assessment Report (2001).

In the USDA and U.S. GHG Inventories, carbon dioxide equivalent (CO_2 eq.) units are expressed in teragrams (Tg), where a teragram equals one million metric tons. The formula for converting gigagrams (1 Gg = 10^9 grams) of a greenhouse gas to teragrams (1 Tg = 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:

$$\mathrm{TgCO_2\ eq.} = (\mathrm{Gg\ of\ gas}) \times (GWP) \times \left({}^{1Tg}\!/_{1{,}000Gg} \right)$$

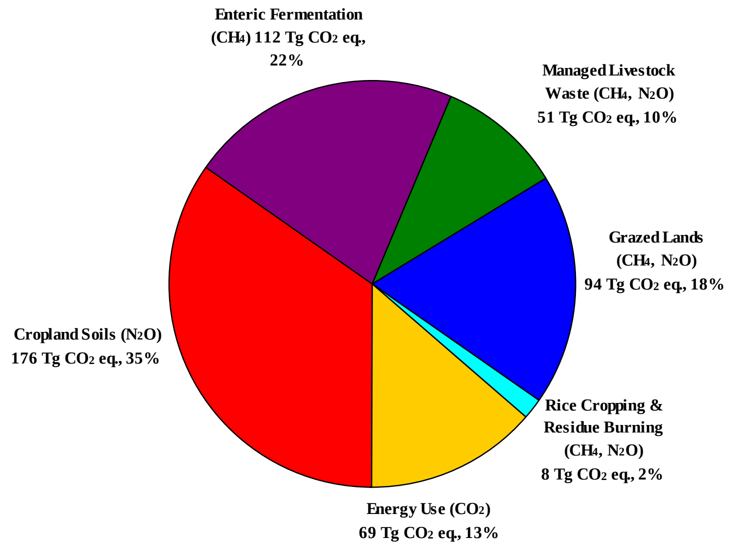
In the land use sector, where carbon dioxide gas is sequestered and stored as carbon (C) in biomass and soils, greenhouse gas removals are often expressed in units of million metric tons of carbon equivalent (MMTCE). The formula below shows how to convert MMTCE to Tg CO_2 eq., and is based on the molecular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

$$\mathrm{TgCO_2\ eq.} = \mathrm{MMTCE} \times \left({}^{44}\!/_{12} \right)$$

10% from managed waste, and 18% from grazed lands. The remaining 13% of total emissions result from agriculturally related energy usage, which is listed under the Energy heading by EPA (2007), 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

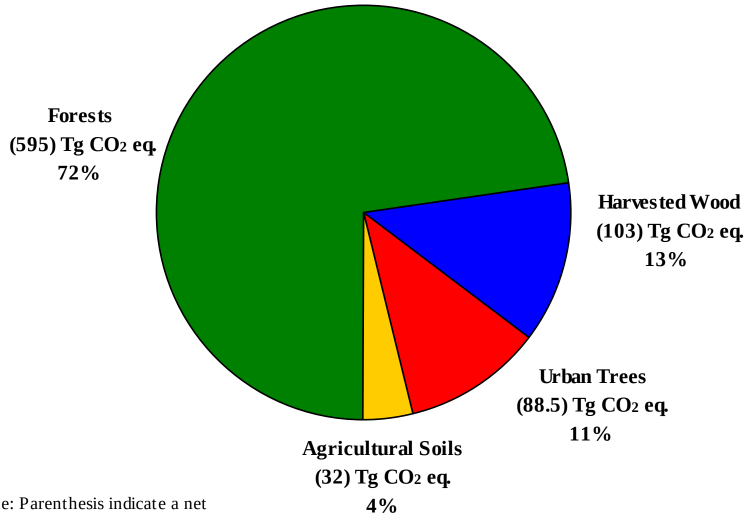
Agricultural Sources of Greenhouse Gas Emissions in 2005



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 2005 (Table 1-2, Figure 1-3). The sink strength of the forest pool has increased 20% since 1990 (Table 1-2, Figure 1-3). Note that cropland soil N_2O emissions reported here are lower than those reported in EPA (2007) because a mistake was found in the calculations reported in EPA (2007). The soil N_2O emissions reported here are consistent with those reported in EPA (2008).

Figure 1-2

Agricultural Sinks of Carbon Dioxide in 2005

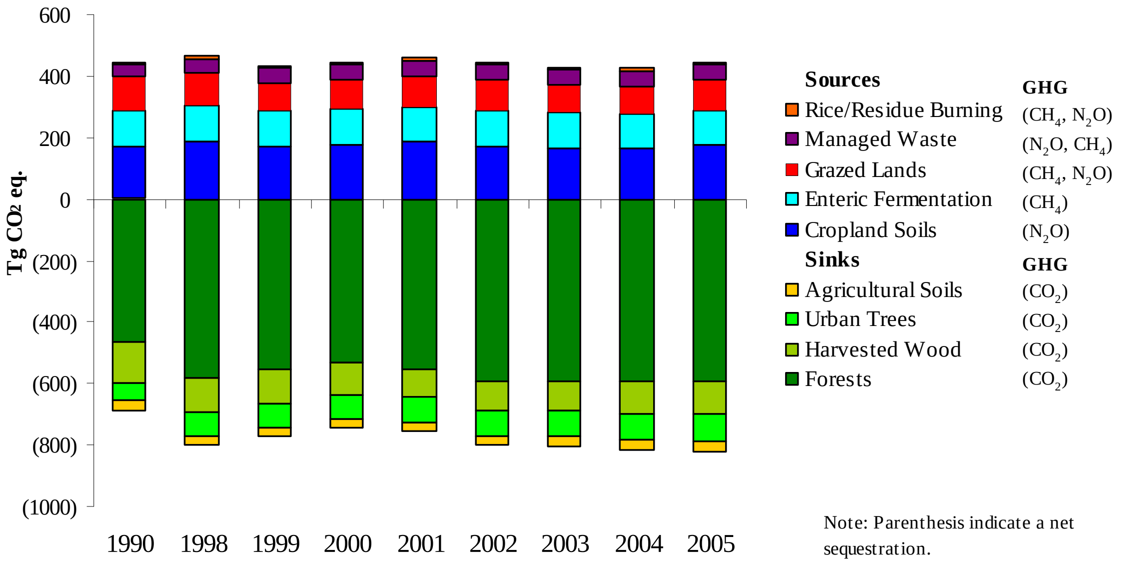


Note: Parenthesis indicate a net sequestration.

Annual CO_2 emissions from onfarm energy use in agriculture are small relative to total energy use across all sectors in the United States. In 2005, fuel and electricity consumption associated with crop and livestock operations resulted in 69 Tg CO_2 (Table 1-1), which is about 1% of overall energy-related CO_2 emissions for 2005 (5943 Tg CO_2). Electricity use led to about 30% of CO_2 emissions from energy use in agriculture; diesel fuel use led to about 46%, while gasoline, natural gas, and liquefied petroleum gas contributed 12%,

Figure 1-3

Agriculture and Forestry Emissions and Offsets for 1990, 1998-2005



Note: Parenthesis indicate a net sequestration.

7%, and 5%, respectively, to total CO_2 emissions from energy use in agriculture.

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 2001). 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 by-products.

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

Table 1-2 Summary of Agriculture and Forestry Emissions and Offsets, 1990, 1998-2005

		1990	1998	1999	2000	2001	2002	2003	2004	2005
Source	GHG	*Tg CO_2 eq.*								
Livestock		**157.4**	**164.6**	**164.3**	**164.0**	**164.6**	**165.5**	**164.9**	**161.8**	**162.9**
Enteric Fermentation	CH_4	117.9	116.7	116.8	115.6	114.6	114.7	115.1	112.6	112.1
Managed Waste	CH_4	30.9	38.7	38.3	38.7	40.1	41.1	40.5	39.7	41.3
Managed Waste	N_2O	8.6	9.2	9.2	9.6	9.8	9.7	9.3	9.4	9.5
Grassland		**96.5**	**104.0**	**88.5**	**93.5**	**102.3**	**101.4**	**89.8**	**89.6**	**96.5**
Grassland	CH_4	2.6	2.7	2.6	2.5	2.5	2.5	2.5	2.5	2.5
Grassland	N_2O	108.4	101.3	85.9	91.0	99.8	99.0	87.5	87.3	94.2
Grassland	CO_2	(14.4)	0.0	0.0	(0.0)	(0.1)	(0.1)	(0.1)	(0.2)	(0.2)
Crops		**157.2**	**169.5**	**154.5**	**158.2**	**163.0**	**148.0**	**143.6**	**142.9**	**153.0**
Cropland Soils[1]	N_2O	168.5	188.3	172.9	178.8	184.9	170.6	166.5	166.1	176.9
Cropland Soils[2]	CO_2	(19.5)	(28.0)	(28.0)	(29.3)	(30.8)	(30.6)	(31.1)	(32.2)	(32.2)
Rice Cultivation	CH_4	7.1	7.9	8.3	7.5	7.6	6.8	6.9	7.6	6.9
Residue Burning	CH_4	0.7	0.8	0.8	0.8	0.8	0.7	0.8	0.9	0.9
Residue Burning	N_2O	0.4	0.5	0.4	0.5	0.5	0.4	0.4	0.5	0.5
Energy Use[3]	CO_2	**44.3**	**57.1**	**60.1**	**53.8**	**73.5**	**52.6**	**44.8**	**52.0**	**69.4**
Forestry		**(656.0)**	**(769.2)**	**(743.7)**	**(716.9)**	**(725.9)**	**(770.4)**	**(771.0)**	**(783.7)**	**(787.2)**
Forests	CO_2	(466.5)	(584.2)	(551.8)	(529.4)	(555.5)	(595.3)	(595.3)	(595.3)	(595.3)
Harvested Wood	CO_2	(132.0)	(111.1)	(115.9)	(109.3)	(90.2)	(92.8)	(91.3)	(101.9)	(103.4)
Urban Trees[4]	CO_2	(57.5)	(74.0)	(76.0)	(78.2)	(80.2)	(82.3)	(84.4)	(86.4)	(88.5)
Net Emissions	***All GHGs***	**(200.6)**	**(274.1)**	**(276.3)**	**(247.4)**	**(222.5)**	**(302.9)**	**(327.9)**	**(337.5)**	**(305.5)**

Note: Parentheses indicate a net sequestration.

[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 CRP program, in addition to cultivated mineral and organic soils.

[3] Includes emissions from electricity use only for 2001 and 2005.

[4] All years except 2001 and 2005 are interpolated values.

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.

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 U.S.

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, inoculating 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 for capturing and using CH_4 emissions from livestock waste 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: 1990-2005 was developed to include emission estimates for years not included in the first U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2001 (USDA 2004) and to revise estimates for previous years based on improved methodologies. This inventory provides a comprehensive assessment of the contribution of U.S. agriculture and forestry to greenhouse gas emissions. 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 2007). 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.

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 Global 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 in-depth 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,
- 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 U.S. and soil N_2O emissions are small because forests do not receive large N additions. 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 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 first 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 Second Edition of the Inventory

This edition includes three major improvements. First, more sophisticated methodologies were used to estimate GHG emissions from cropped and grazed soils. Second, the livestock chapter now includes emissions from grazed soils that were previously included in the cropland chapter. Lastly, this report includes more quantitative estimates of uncertainty and mitigation potential. The first edition qualitatively discussed uncertainty and mitigation but quantitative analyses were limited. Similarly, the first edition included little quantification of mitigation potential, which is now included in chapters 2 and 3. In addition to updating GHG flux estimates for 1990-2001, estimates for 2002-2005 are included.

The first edition of the USDA GHG inventory estimated GHG emissions and sinks from non-rice crops and grazed lands based solely on IPCC (1997) Tier 1 methodology. Instead of relying exclusively on IPCC (1997) Tier 1 methodology for these sources, the current inventory uses the CENTURY and DAYCENT ecosystem models to simulate GHG fluxes for cropped and grazed lands. Use of more sophisticated process based models is known as an IPCC Tier 3 methodology. The 2005 EPA GHG inventory was the first to use a process-based model (DAYCENT) to estimate N_2O emissions and the 2006 EPA inventory was the first to use a process-based model (CENTURY) to estimate CO_2 fluxes. Tier 1 IPCC (1997) methodology has traditionally been used to estimate U.S. GHG fluxes, although other higher tier methods which have been demonstrated to accurately represent GHG emissions and sequestration are encouraged by IPCC's guidelines. The major advantages of the Tier 1 methodology are ease of implementation and high degree of transparency. GHG flux estimates are based on simple empirical relations that can easily be implemented using spreadsheets. For example, IPCC (2006) Tier 1 methodology assumes that 1.0% of the nitrogen in fertilizer added to soils is emitted directly as N_2O from soils on an annual basis. The method also accounts for nitrogen that is added to cropped soils but is removed either by volatilization or leaching and deposited elsewhere by prescribing an emission factor of 1% and 0.75%, respectively. The disadvantage of this method is that other factors which influence emissions (e.g., soil type, weather, previous land use) are not accounted for or are only included in a rudimentary manner. To more realistically account for these other factors, simulation models have been developed that can be applied at large scales to estimate GHG fluxes. More advanced methods which use simulation models should yield more reliable estimates because they account for many of the factors that influence emissions but are not included in IPCC (2006) Tier 1 methodology. The disadvantage of using simulation models for regional and national assessments is that considerable computational power and programming expertise are required to perform large-scale simulations. Additionally, large amounts of time and data are required to acquire and format model inputs and to test the reliability of model outputs. This is why these types of models have not been used for national assessments until recently.

Another major change relates to emissions from livestock production. The livestock chapter is now entitled “Livestock and Grazed Land Emissions.” In the first edition, carbon stock changes for grazed lands were included with Cropland Agriculture. However, carbon fluxes for grazed lands are now included in the livestock chapter (Chapter 2) so that all fluxes associated with livestock production are attributed to the livestock sector.

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Chapter 2: Livestock and Grazed Land Emissions

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

A total of 259 Tg CO_2 eq. of greenhouse gases (GHGs) were emitted from livestock, managed livestock waste, and grazed land in 2005 (Table 2-1, Figure 2-1). This represents about 49% of total emissions from the agricultural sector (EPA 2007). Compared to the baseline year (1990), emissions from this source were about 2% lower in 2005. The 95% confidence interval for 2005 was estimated to lie between 239 and 306 Tg CO_2 eq. (Table 2-1).

Enteric fermentation was responsible for almost half (112 Tg CO_2 eq.) of all emissions associated with livestock production, while grazed lands (96 Tg CO_2 eq.) and managed waste (50 Tg CO_2 eq.) accounted for approximately 40% and 20% of the total emissions. All of the emissions from enteric fermentation and about 81% of emissions from managed livestock waste were in the form of methane (CH_4). Of the emissions from grazed lands, 97% were in the form of nitrous oxide (N_2O) (Table 2-2). 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 were roughly neutral for CO_2 emissions in 2005 (Table 2-2). The largest total emissions associated with livestock production were from 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.

Table 2-1 Greenhouse Gas Emission Estimates and Uncertainty Intervals in 2005

	Estimate	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Source		*Tg CO_2 eq.*			*%*
CH_4 enteric fermentation	112	100	132	(11)	18
CH_4 managed waste + grazed land	43	35	52	(18)	20
N_2O managed waste	10	8	12	(16)	24
N_2O grazed land	94	82	136	(13)	44
CO_2 grazed land remaining grazed land	16	13	18	(18)	15
CO_2 land converted to grazed land	(16)	(18)	(14)	(13)	14
Total	**259**	**239**	**306**	**(8)**	**18**

Beef cattle were responsible for the largest fraction (65%) of GHG emissions from livestock in 2005, 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 (20%), primarily CH_4 from enteric fermentation and managed waste. The third largest GHG source from livestock was swine (8%), 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 2005

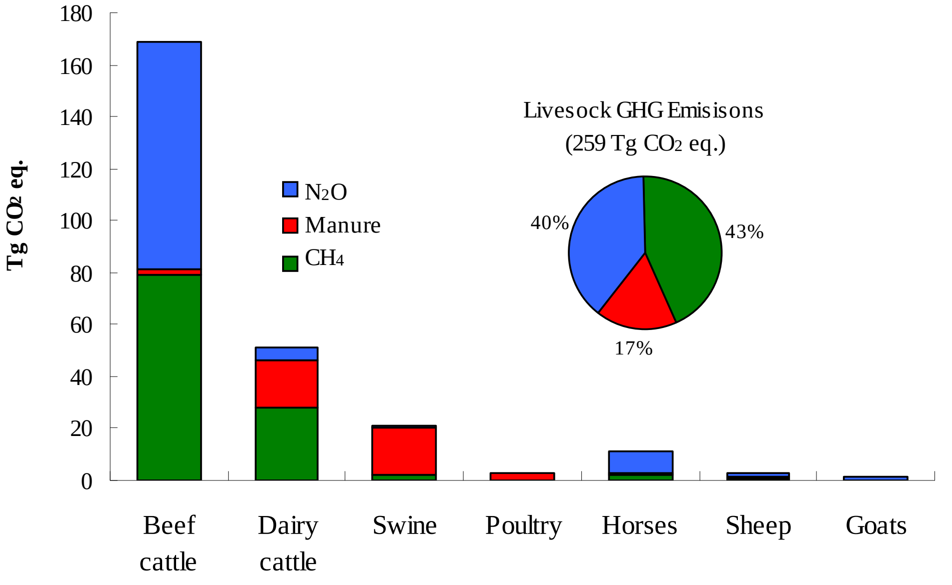
	Enteric Fermentation	Managed Livestock Waste		Grazed Land			
	CH_4	CH_4	N_2O	N_2O[1]	CH_4	CO_2	***Total***
Animal Type			*Tg CO_2 eq.*				
Beef cattle	79.22	0.41	5.78	81.14	1.91	(0.19)	**168.3**
Dairy cattle	27.69	18.75	2.52	2.25	0.00	(0.01)	**51.2**
Swine	1.92	18.65	0.48	0.00	0.00	0.00	**21.0**
Horses	2.00	0.00	0.20	8.30	0.47	(0.02)	**11.0**
Poultry	0.00	2.66	0.44	0.00	0.00	0.00	**3.1**
Sheep	1.03	0.00	0.07	1.73	0.08	(0.00)	**2.9**
Goats	0.30	0.00	0.02	0.78	0.02	(0.00)	**1.1**
Total	**112.2**	**40.5**	**9.5**	**94.2**	**2.5**	**(0.2)**	**258.6**

Note: Parenthesis indicate a net sequestration.

[1] Includes direct and indirect emissions.

Figure 2-1

Greenhouse Gas Emissions from Livestock in 2005



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 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 grazed soils 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. Nitrous oxide emissions from managed livestock waste applied to cropped soils are included in the Cropland Agriculture chapter, although emissions associated with waste applied to grazed land are included in this chapter. State-level livestock population data also are presented in this chapter because GHG emissions from livestock are related to livestock population sizes.

In contrast to the first edition of the USDA GHG report (USDA 2004) that relied exclusively on IPCC (1997) methodology, this edition includes estimates for N_2O emissions and CO_2 fluxes from grazed land obtained from the DAYCENT and CENTURY ecosystem models. Another change compared to the first edition is that carbon (C) stock changes in grazed lands that were previously included in the Cropland Agriculture chapter are now included in this chapter.

2.2 Sources of Greenhouse Gas Emissions from Livestock

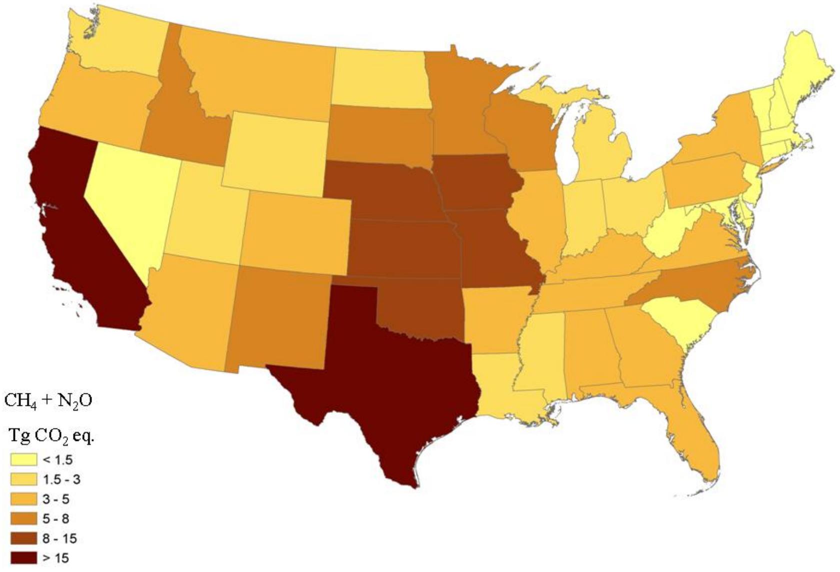
The mechanisms and important factors in generating GHG fluxes from livestock, waste management, and grazed lands are detailed below.

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 by-product. 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

Map 2-1

GHG Emissions from Livestock in 2005



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 is "unmanaged" when it is deposited directly on grazed lands. Alternatively, livestock waste can be "managed" in storage and treatment systems, or spread on fields in lieu of long-term storage. Many livestock producers in the U.S. manage livestock waste in systems such as solid storage,

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

		Relative Emissions	
Manure Management System	**Description**	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 unine are collected and transported in a liquid state to tanks for storage. The liquid/slurry mixture may be storaed 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 systems and transported to lagoons for storage. Manure/urine reside 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	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.

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

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.

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 CH_4 is not 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, while 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, 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 runoff 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 runoff into waterways can be converted to N_2O by aquatic denitrification.

2.2.3 Grazed Lands

Nitrous oxide from soils is the primary GHG gas 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) (USDA 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 2005 are provided in Appendix Table A-1. Appendix Table A-2 shows total national livestock population sizes from 1990 to 2005 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 just over 14 million head in 2005 (Appendix Table A-1). Kansas, Nebraska, Oklahoma, and Missouri each raised from 6 to 4 million head of beef cattle, while several other States raised ~2 million head. Fewer dairy cattle than beef cattle are raised in the United

States. Dairy cattle populations were highest in California (~2.4 million) and Wisconsin (~1.9 million) (Appendix Table A-1). Minnesota, New York, and Pennsylvania had the next largest populations of dairy cattle, ranging from 730,000 to 940,000 head in each State. Most States had fewer than 500,000 head of dairy cattle.

Iowa was the largest swine producer with 16 million head in 2005 (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 2005, with roughly 260 million head of poultry in each State (Appendix Table A-1). Alabama, North Carolina, Mississippi, and Texas also had large populations of poultry, between 138 and 205 million head each. Indiana, Kentucky, Maryland, Oklahoma, and Virginia had poultry populations between 50 and 60 million head.

2.4 Enteric Fermentation

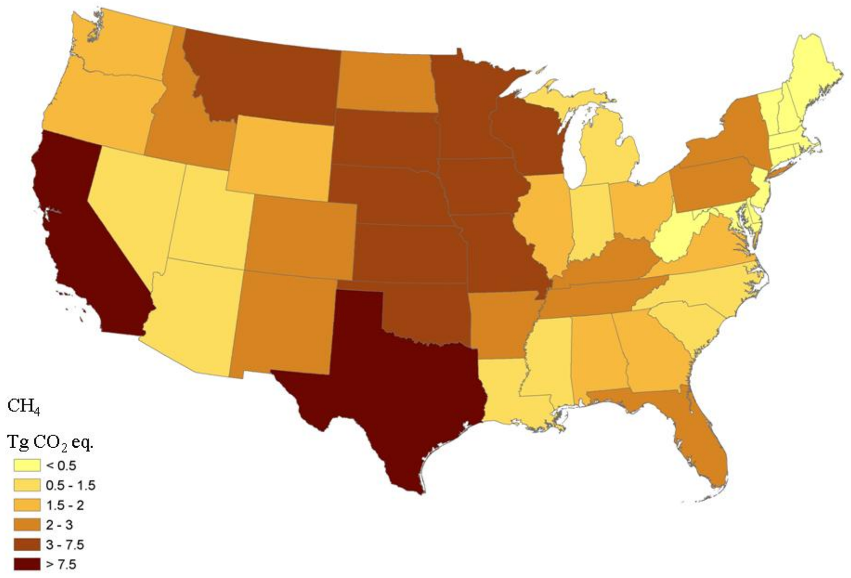
Just about half (43%) of emissions associated with livestock production were from CH_4 produced by enteric fermentation. Cattle were responsible for the vast majority of enteric CH_4 emissions (95%) in 2005 (Table 2-2). Texas (14.4 Tg CO_2 eq.) and California (7.8 Tg CO_2 eq.) had the largest CH_4 emissions from enteric fermentation across all livestock types in 2005 (Map 2-2, Appendix Table A-3, 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). Central, Northern Plains, and some Great Lakes States also had relatively high CH_4 emissions from enteric fermentation, ranging between 3 and 7.5 Tg CO_2 eq. per State in 2005. 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.

Annual emissions of CH_4 from enteric fermentation fluctuated up and down by less than approximately 10 Tg CO_2 eq. between 1990 and 2005 (Table 2-4). Emissions peaked in 1995 and then decreased by about 10 Tg CO_2 eq. by 2005 (~9% of total). Overall, by 2005, CH_4 emissions from enteric fermentation declined by about 4% compared to 1990 levels.

2.4.1 Methods for Estimating Methane Emissions from Enteric Fermentation

Map 2-2

Methane Emissions from Enteric Fermentation in 2005



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 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 model based on recommendations provided in IPCC (1997) and IPCC (2000) 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

Table 2-4 U.S. Methane Emissions from Enteric Fermentation in 1990, 1995-2005

	1990	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Animal Type						*Tg CO_2 eq.*						
Beef cattle	83.2	89.7	88.8	86.6	85.0	84.9	83.4	82.5	82.4	82.6	80.4	79.2
Dairy cattle	28.9	27.7	26.3	26.4	26.3	26.6	27.0	26.9	27.1	27.3	27.0	27.7
Horses	1.9	1.9	1.9	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0
Sheep	1.9	1.5	1.4	1.3	1.3	1.2	1.2	1.2	1.1	1.1	1.0	1.0
Swine	1.7	1.9	1.8	1.8	2.0	1.9	1.9	1.9	1.9	1.9	1.9	1.9
Goats	0.3	0.2	0.2	0.2	0.2	0.2	0.3	0.3	0.3	0.3	0.3	0.3
Total	**117.9**	**123.0**	**120.5**	**118.3**	**116.7**	**116.8**	**115.6**	**114.6**	**114.7**	**115.1**	**112.6**	**112.1**

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 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 2002a, 2001a, 2000a, 1999a, 1995, Livestock slaughter: USDA NASS 2002b, 2001b, 2000b). Beef calf birth percentages were obtained from the USDA APHIS National Animal Health Monitoring System (USDA APHIS NAHMS 1998, 1994, 1993).

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), 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 (1996). 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 (2000).

Step 3: Estimate Methane Emissions from Cattle

Emissions were estimated in three steps: a) determine gross energy intake using the IPCC (2000) equations, b) determine an emissions factor using the GE values and other factors, and c) sum the daily emissions for each animal type. The necessary data values include:

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[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.

- 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)
- 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 ten 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 reports (Hogs and pigs: USDA NASS 2002c, 2001c, 2000c, 1999b, 1998, 1994a, Sheep and goats: USDA NASS 2002d, 2001d, 2000d, 1999c, 1994b). 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 2002). 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 2007) 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 are ±10% or lower. However, the uncertainty for many of the emission factors are over ±20%. The overall 95% confidence interval around the estimate of 112 Tg CO_2 eq. ranges from 100 to 132 Tg CO_2 eq. (Table 2-1).

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 2005, accounting for 81% of 50 Tg CO_2 eq. total emissions from this source (Table 2-5). The remaining 19% of GHG emissions from managed livestock waste was N_2O. Dairy cattle and swine were each responsible for approximately 40% of total managed waste emissions (Figure 2-2). Poultry (6%) and beef cattle (16%) were also important sources in 2005. For beef cattle, N_2O was the predominate form (71%) of waste emissions. Over time, emissions from managed waste increased by ~28% from 1990 to 2005 (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.

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 2005.

Managed manure emissions from horses, sheep, and goats are small due to the relatively small population of these animals (Appendix Table A-2), as for beef cattle, most of the manure is unmanaged or managed in dry systems (EPA 2007).

Table 2-5 Greenhouse Gas Emissions from Managed Livestock Waste in 1990, 1995-2005

	1990	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
GHG Type						*$TgCO_2$ eq.*						
Nitrous Oxide[1]	8.6	9.0	8.7	9.0	9.2	9.2	9.6	9.8	9.7	9.3	9.4	9.5
Methane[2]	30.9	35.1	33.7	35.4	38.7	38.3	38.7	40.1	41.1	40.5	39.7	41.3
Total	**39.5**	**44.1**	**42.4**	**44.4**	**47.9**	**47.5**	**48.3**	**50.0**	**50.8**	**49.8**	**49.2**	**50.8**

[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.

Figure 2-2

Greenhouse Gas Emissions from Managed Livestock Waste by Livestock Type in 2005

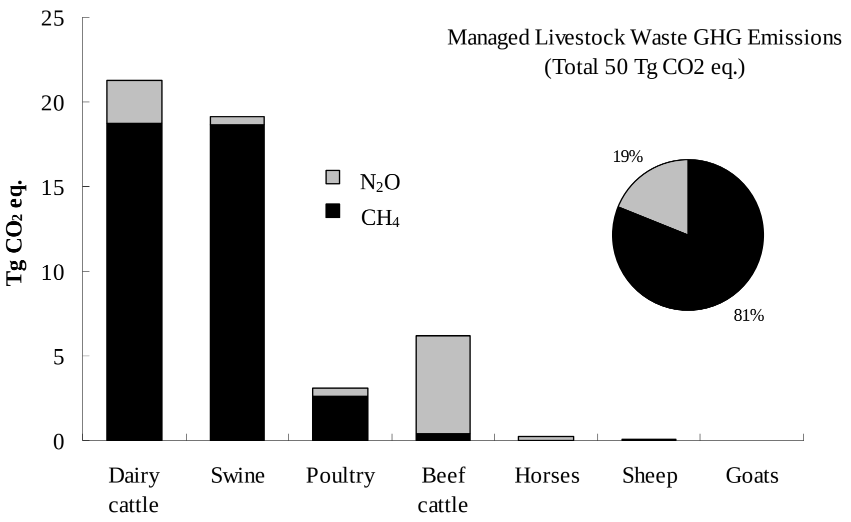
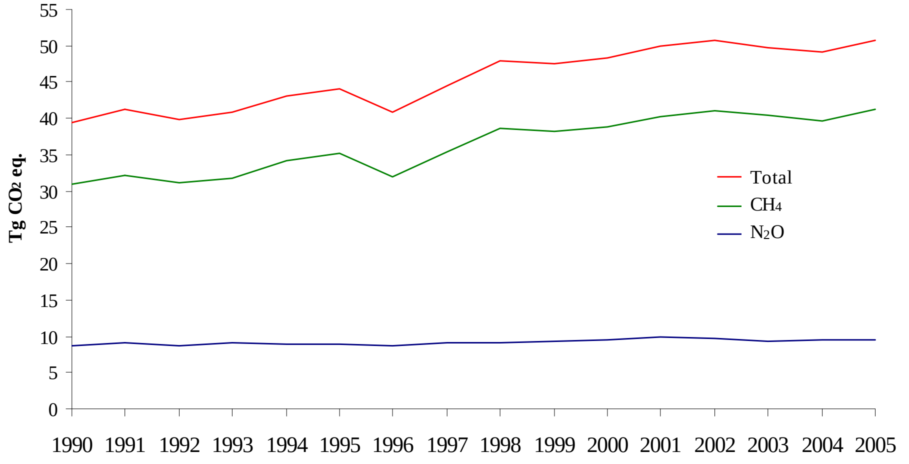


Figure 2-3

Greenhouse Gas Emissions from Managed Livestock Waste, 1990-2005



State-level GHG emissions from managed livestock waste varied across States in 2005, 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 (7 and 6 Tg CO_2 eq., respectively) (Appendix Table A-13, Map 2-3). In California, GHG emissions from managed livestock waste were largely from dairy cattle, while in Iowa, they were largely from swine (Appendix Table A-14, A-15). North Carolina and Texas also had large GHG emissions from managed livestock waste (4 and 3 Tg CO_2 eq., respectively). 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).

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 2007) as well as for this inventory report. Animal population data is used to estimate CH_4 production potential and nitrogen in waste, and these are multiplied by a methane conversion factor (MCF) and an N_2O emission factor. 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. Nitrous oxide emission factors are determined by State and livestock type. 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 2007).

Data required to calculate emissions from livestock waste:

- State-level animal population data by animal type
- Animal type specific nitrogen excretion rate
- Animal type specific volatile solid production
- Animal type specific CH_4 production potential
- Extent CH_4 production potential is realized (including biogas collection efforts)
- State-level portion of manure in each management system by animal type
- Portion of manure deposited on grasslands and used in spread operations

Seven animal 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 1996a, 1998a, 2000a, 2000b, 2000c) and EPA (ERG 2000, EPA 2002a, 2002b). For beef cattle and poultry, manure management system usage is not tied to farm size and is based on other sources (ERG 2000, USDA 2000d, UEP 1999). For other animal types, manure management system usage is based on previous estimates (EPA 1992a).

Methane and N_2O emissions calculations are based on the following animal characteristics for each relevant livestock population:

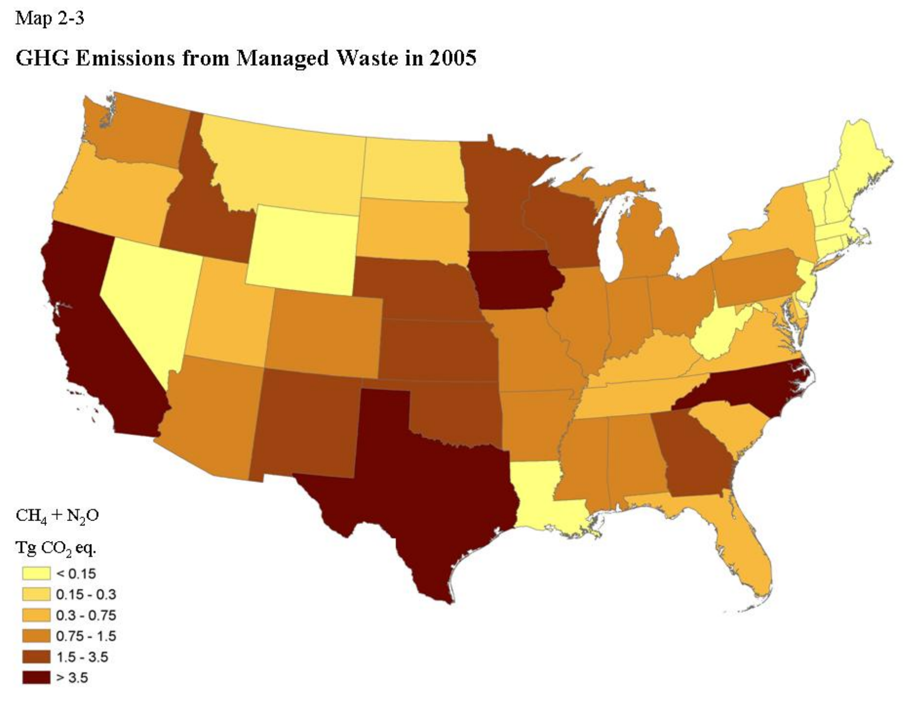
- Volatile solids excretion rate (VS)
- Maximum CH_4 producing capacity (B_o) for U.S. animal waste
- Nitrogen excretion rate (N_{ex})
- Typical animal mass (TAM)

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.

Methane conversion factors for dry manure management systems and N_2O emissions factors for all management systems were set equal to the default IPCC factors for temperate climates (IPCC 2000). 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.

Nitrous oxide emissions were estimated by multiplying total Kjeldahl nitrogen (TKN) production for livestock waste by State-specific emission factors. TKN was calculated for each animal type using national average nitrogen excretion rate (USDA 1996a). N_2O emission factors were weighted by State-level types of manure management.


Map 2-3

GHG Emissions from Managed Waste in 2005

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 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 35 to 52 Tg CO_2 eq. around the inventory value of 43 Tg CO_2 eq., and the manure management N_2O inventory has a 95% confidence interval from 8 to 12 Tg CO_2 eq. around the inventory value of 10 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

Table 2-6 Greenhouse Gas Emissions from Grazed Lands in 1990, 1995-2005

	1990	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
GHG Type						*$TgCO_2$ eq.*						
Nitrous Oxide[1]	**108.4**	**102.1**	**120.9**	**98.3**	**101.3**	**85.9**	**91.0**	**99.8**	**99.0**	**87.5**	**87.3**	**94.2**
Direct	88.0	77.8	96.5	76.3	79.9	68.9	73.9	74.8	80.1	71.0	71.3	76.4
Indirect Volatilization	10.7	10.3	10.2	10.1	10.1	9.6	9.3	9.4	9.3	9.4	9.1	9.9
Indirect Leaching & Run-Off	9.6	14.0	14.3	11.9	11.2	7.4	7.8	15.7	9.5	7.1	6.9	7.9
Methane[2]	**2.6**	**2.7**	**2.7**	**2.6**	**2.7**	**2.6**	**2.5**	**2.5**	**2.5**	**2.5**	**2.5**	**2.5**
Carbon Dioxide	**(14.4)**	**0.1**	**0.1**	**0.1**	**0.0**	**0.0**	**(0.0)**	**(0.1)**	**(0.1)**	**(0.1)**	**(0.2)**	**(0.2)**
Grazed Lands Remaining Grazed	0.1	16.4	16.4	16.4	16.4	16.3	16.3	16.2	16.2	16.2	16.1	16.1
Land Convertd to Grazed Land	(14.6)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)	(16.3)
Total	**96.5**	**104.9**	**123.6**	**101.0**	**104.0**	**88.5**	**93.5**	**102.3**	**101.4**	**89.8**	**89.6**	**96.5**

[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.

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.

2.6 Grazed Lands

Grazed lands 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 a source or a sink of CO_2.

Nitrous oxide was the predominant GHG emitted from grazed lands in 2005, accounting for 98% of all emissions from this source (Table 2-6). The remaining 2% of GHG emissions from grazed lands was CH_4. Grazed lands were roughly CO_2 neutral in 2005, with a small uptake of 0.2 Tg CO_2 eq. through sequestration of CO_2 in soil organic carbon. Nitrous oxide emissions from grazed land totaled 94 Tg CO_2 eq. in 2005 (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 managed grazed land were about twice those of managed manure in 2005 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).

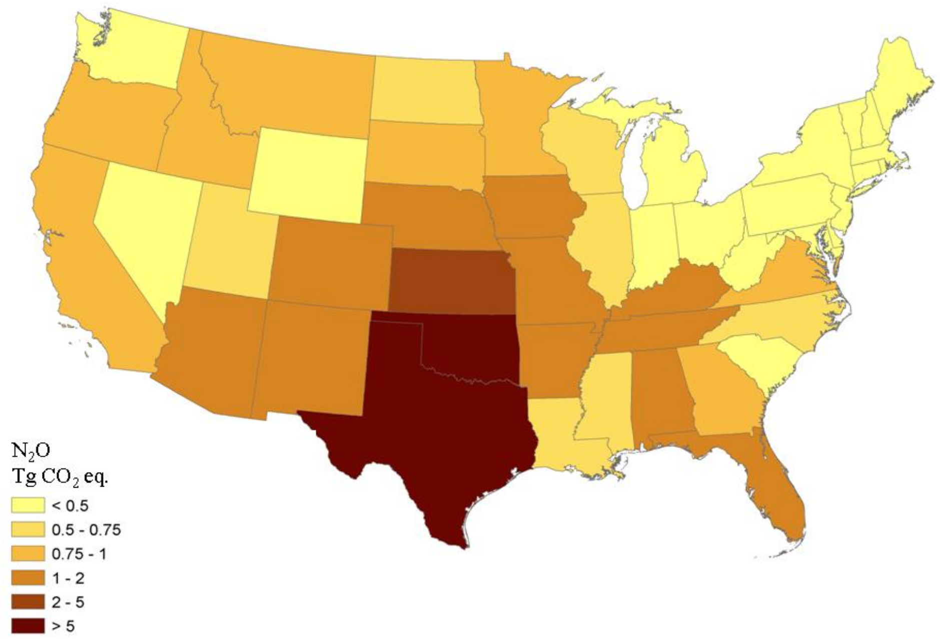
2.6.1 Methodology To Estimate Nitrous Oxide Emissions from Grazed Lands

Estimates of N_2O emissions from this component were based on DAYCENT model simulations of grazed lands, estimates of animal waste production (Appendix Table A-21), and IPCC (2006) methodology for emissions associated with nitrogen from unmanaged manure not accounted for by the DAYCENT simulations (Del Grosso et al. 2006). Unmanaged manure is not managed in manure management systems, but instead is deposited directly on soils 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.

The DAYCENT ecosystem model simulated improved pastures and rangelands at county-level resolution for the U.S. Improved pastures are defined as grazing lands that were seeded with legumes and/or were 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.

Map 2-4

Nitrous Oxide Emissions from Grazed Land in 2005



Key model inputs are daily weather, soil texture class, vegetation mix, 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, animal waste nitrogen production, and legume fixation. The DAYCENT simulations are described in more detail in Chapter 3 of this report and in EPA (2007) and Del Grosso et al. (2006).

Comparisons of animal waste nitrogen production with estimates based on animal numbers show that DAYCENT did not account for 100% of animal waste nitrogen. IPCC (2006) methodology was applied to estimate emissions for the nitrogen inputs from this source not accounted for by the DAYCENT simulations. IPCC methodology was also used to estimate indirect emissions from DAYCENT simulated nitrogen volatilization and NO_3 leaching. IPCC (2006) methodology and details on how animal populations, manure, and nitrogen in waste production data were acquired are described in detail in Appendix 3.11 of the U.S. GHG Inventory (EPA 2007). 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.

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 are all outputs for the grazed lands simulated by DAYCENT. For animal waste not accounted for by the DAYCENT simulations, 20% 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. 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 eight cropping experiments in North America. IPCC (2006) methodology was used to estimate uncertainties for the grazed land 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 94 Tg CO_2 eq. for grazed soil N_2O emissions was 82 to 136 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 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 has been reported every 5 years starting in 1982 and 1997 is the most recent year that has been reported. According to NRI data, ~32 million ha of grassland (out of a total ~228 million ha reported in 1997) were converted to grassland between 1993 and 1997. An example of land converted to grassland is land that was cropped historically but then placed in the Conservation Reserve Program. 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 2007).

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 (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.

2.6.5 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 (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 13 to 18 around the estimate of 16 Tg CO_2 eq. in 2005 for land remaining grazed land and (18) to (14) around the estimate of (16) Tg CO_2 eq. for land converted to grazed land in 2005 (Table 2-1).

2.7 Mitigating Greenhouse Gas Emissions from Livestock

2.7.1 Enteric Fermentation

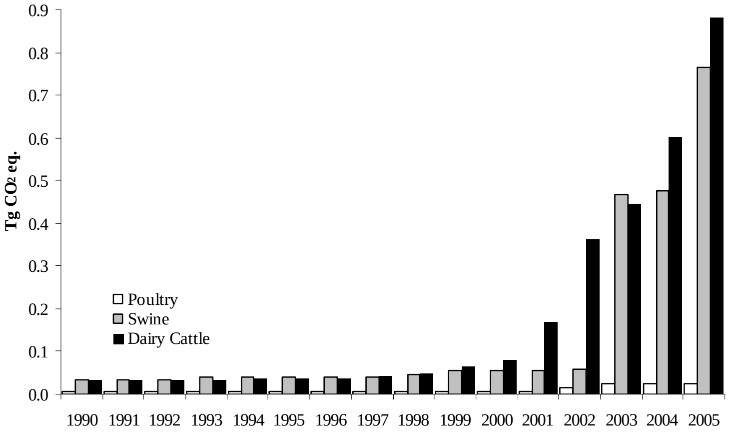
Emissions of CH_4 from enteric fermentation in ruminant and non-ruminant animals are dependent on the animal’s digestive system and the amount and type of feed consumed. On average, beef and dairy cattle convert 6% of gross energy intake from feed into CH_4 through enteric fermentation, constituting 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. 1998b):

- Increasing the digestibility of forages and feeds;
- Providing feed additives which may tie up hydrogen in the rumen;
- Inhibiting the formation of CH_4 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:

Figure 2-4

Estimated Reductions in Methane Emissions from Anaerobic Digesters, 1990-2005



- 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 myriad 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 does produce 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.

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 form biological nitrogen fixation because they are seeded with legumes. This mitigation potential has not been quantified but it will be in future DAYCENT model simulations. Although grazed mineral soils are a net sink of $CO_{2,}$ grazed organic soils are a net source. If half of the grazed organic soils were converted back to wetlands, CO_2 emissions from this source could be reduced from approximately 4.6 to 2.3 Tg CO_2 eq. per year. However, the saturated soil conditions characteristic of wetlands would cause an increase in soil CH_4 emissions and it is unclear to what extent this would nullify reduced CO_2 emissions.

Grazed lands are currently roughly GHG neutral for CO_2 emissions (Table 2-6). However, grazed lands in the U.S. have the potential to store over 100 Tg CO_2 per year (Follett et al. 2001). The largest potential is 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.

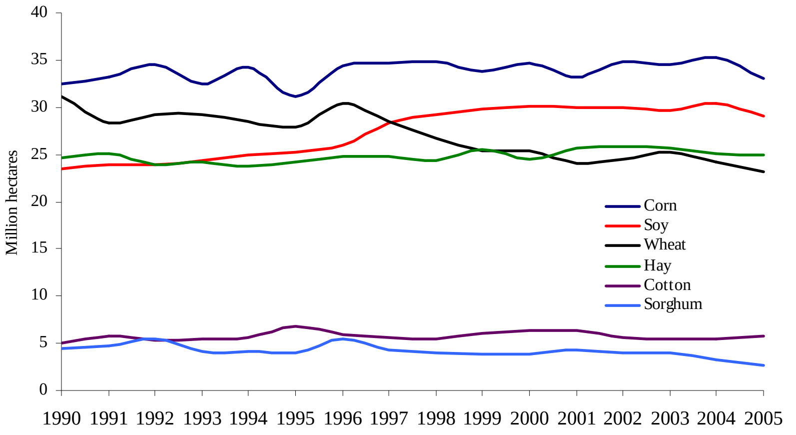
3.1 Summary of U.S. Greenhouse Gas Emissions from Cropland Agriculture

In 2005, cropland agriculture resulted in total emissions of 219.5 Tg CO_2 eq. of greenhouse gases (GHG) (Table 3-1). Cropland agriculture is responsible for about half (53%) of all emissions from the agricultural sector (EPA 2007). Nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4) emissions from agricultural soils totaled 177, 34, and 8 Tg CO_2 eq., respectively, in 2005. However, that amount was offset by a storage, or carbon sequestration, of 66.5 Tg CO_2 eq. in agricultural soils in 2005. Thus, when this is taken into account, net emissions of GHG from cropland agriculture amount to approximately 153 Tg CO_2 eq. The 95% confidence interval for net emissions in 2005 is estimated to lie between 137 and 188 Tg CO_2 eq. (Table 3-1).

Emissions in 2005 were only 4% higher than the baseline year (1990). Greenhouse gas emissions from agricultural soils fluctuated between 1990 and 2005 with no clear trend of increasing or decreasing (Table 3-2). Annual fluctuations are primarily a result of variability in weather patterns and land use changes.

Figure 3-1

U.S. Planted Cropland Area by Crop Type, 1990-2005



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 (Tables 3-1, 3-2). Soil CO_2 emissions from cultivation of organic soils (14%) and from liming (2%) are the remaining sources. Nitrous oxide emissions from soils

Table 3-1 Estimates and Uncertainties for Cropland Greenhouse Gas Emissions, 2005

	GHG Emissions	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Source		*Tg CO_2 eq.*			%
N_2O	**177.4**	**159.8**	**220.8**	**-10**	**24**
Soils	**168.4**	**151.8**	**205.8**	-10	22
Managed Manure [1]	8.5	2.6	30.6	-70	259
Residue Burning	0.50	0.45	0.57	-10	14
CH_4	**7.7**	**3.0**	**19.5**	**-61**	**152**
Residue Burning	0.90	0.75	0.97	-17	8
Rice Cultivation	6.90	2.10	18.60	-70	170
CO_2	**(32.2)**	**(49.7)**	**(16.9)**	**-55**	**47**
Mineral Soils [2]	(66.5)	(77.9)	(55.2)	-17	17
Organic Soils	30.3	18.4	39.6	-39	31
Liming of Soils	4.0	0.2	8.0	-96	98
Total Emissions	**219.5**	**197.4**	**265.6**	**-10**	**21**
Net Emissions [3]	**153.0**	**136.6**	**187.9**	**-11**	**23**

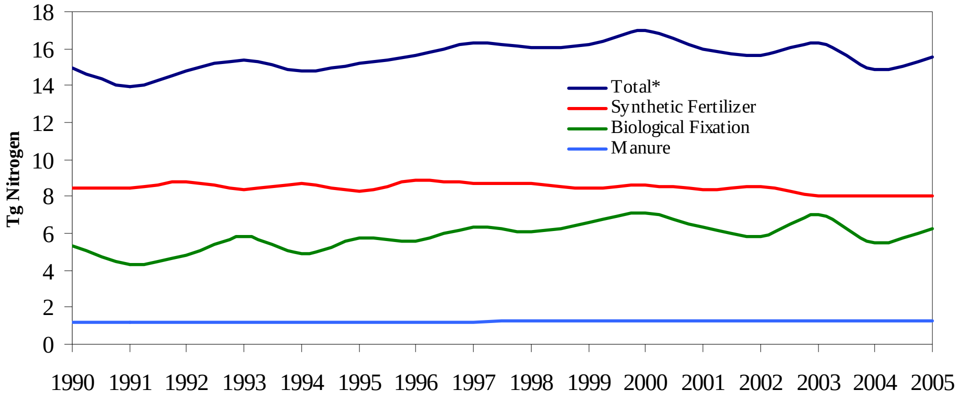
[1] Accounts for loss of manure N during transport, treatment and storage, including both volatilization and leaching/runoff.

[2] Soil carbon sequestration on land under the Conservation Reserve Program and soil carbon fluxes for land converted to cropland are included with mineral soils.

[3] Includes sources and sinks.

Figure 3-2

Annual Nitrogen Inputs to Cropland Soil



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

are the largest source in the U.S. due to the fact that 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. 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 more cropland has been enrolled in the Conservation Reserve Program (CRP).

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 emissions followed by soybean and wheat (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

Table 3-2 Summary of Greenhouse Gas Emissions from Cropland Agriculture, 1990, 1998-2005

	1990	1998	1999	2000	2001	2002	2003	2004	2005
Source					*Tg CO_2 eq.*				
N_2O	**168.9**	**188.8**	**173.4**	**179.2**	**185.4**	**171.0**	**167.0**	**166.6**	**177.4**
Soils	161.0	180.1	164.7	170.4	176.5	161.9	158.0	157.7	168.4
Managed Manure[1]	7.5	8.2	8.3	8.4	8.5	8.7	8.5	8.5	8.5
Residue Burning	0.4	0.5	0.4	0.5	0.5	0.4	0.4	0.5	0.5
CH_4	**7.8**	**8.7**	**9.1**	**8.3**	**8.4**	**7.5**	**7.7**	**8.4**	**7.7**
Residue Burning	0.7	0.8	0.8	0.8	0.8	0.7	0.8	0.9	0.9
Rice Cultivation	7.1	7.9	8.3	7.5	7.6	6.8	6.9	7.6	6.9
CO_2	**(19.5)**	**(28.0)**	**(28.0)**	**(29.3)**	**(30.8)**	**(30.6)**	**(31.1)**	**(32.2)**	**(32.2)**
Mineral Soils[2]	(54.0)	(63.0)	(62.8)	(64.0)	(65.6)	(65.8)	(66.0)	(66.4)	(66.5)
Organic Soils	29.8	30.3	30.3	30.3	30.3	30.3	30.3	30.3	30.3
Liming of Soils	4.7	4.7	4.5	4.3	4.4	5.0	4.6	3.9	4.0
Total Emissions	**211.2**	**232.5**	**217.2**	**222.2**	**228.5**	**213.9**	**209.5**	**209.3**	**219.5**
Net Emissions	**157.2**	**169.5**	**154.5**	**158.2**	**163.0**	**148.0**	**143.6**	**142.9**	**153.0**

Note: Parenthesis indicate a net sequestration.

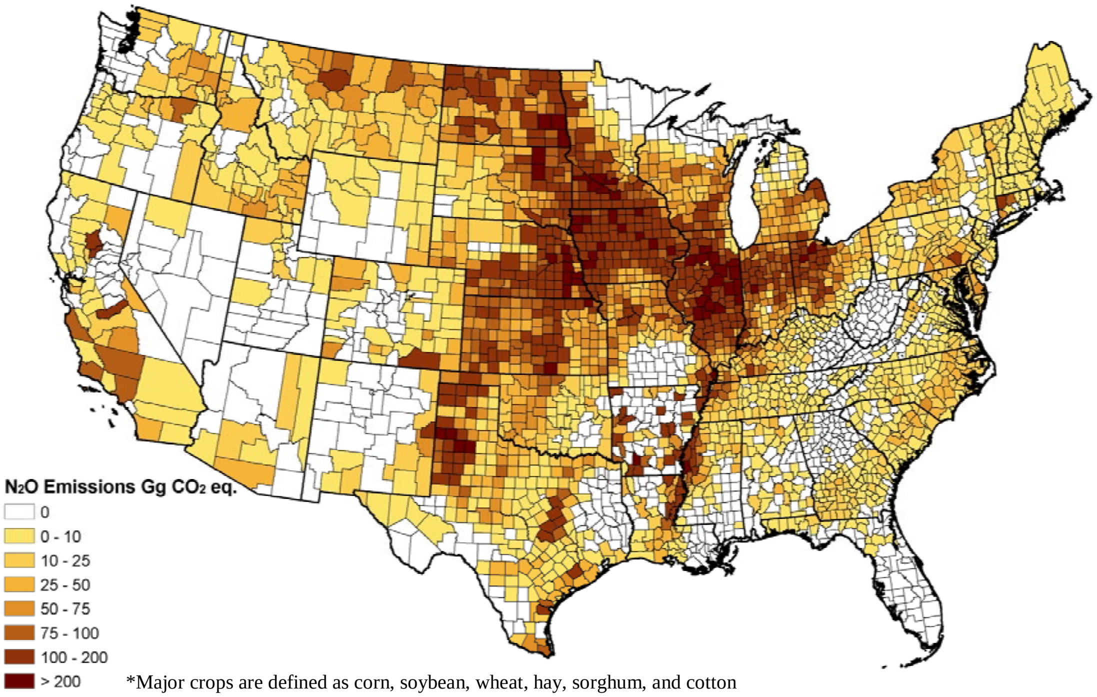
[1] Accounts for loss of manure N during transport, treatment and storage, including both volatilization and leaching/runoff.

[2] Soil C sequestration on CRP land and soil C fluxes for land converted to cropland are included with mineral soils.

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. Synthetic fertilizer makes up about half of total N additions, followed by fixation and manure (Figure 3-2). Note that Map 3-1 does not include emissions from non-major crops, which make up a significant potion of total emissions in California and Florida. Soil N_2O emissions reported here are lower than those reported in EPA (2007) because a mistake was found in the calculations reported in EPA (2007). The cropped soil emissions reported here are consistent with those in EPA (2008).

Map 3-1

County-Level Nitrous Oxide Emissions from Major Cropped Soils in 2005 *



*Major crops are defined as corn, soybean, wheat, hay, sorghum, and cotton

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 lead 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.

Table 3-3 Nitrous Oxide Emissions from Differently Cropped Soils, 1990-2005[1]

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Source	*Tg CO_2 eq.*															
Corn	**56.6**	**55.0**	**54.6**	**59.2**	**54.1**	**50.3**	**61.5**	**57.2**	**63.9**	**56.6**	**61.1**	**63.4**	**58.2**	**51.9**	**54.1**	**56.5**
Direct	44.2	45.6	44.9	49.0	47.5	41.7	51.4	50.2	53.2	48.5	52.0	53.4	50.0	42.4	47.5	47.5
Volatilization	1.7	1.6	1.8	1.6	1.7	1.6	1.7	1.7	1.8	1.7	1.8	1.7	1.8	1.7	1.7	1.8
Leaching & Runoff	10.7	7.8	7.9	8.7	4.9	7.0	8.4	5.3	8.9	6.4	7.3	8.3	6.3	7.9	4.9	7.3
Soybean	**29.4**	**28.3**	**27.6**	**30.4**	**28.2**	**28.6**	**33.4**	**32.7**	**39.3**	**33.3**	**39.1**	**40.8**	**36.2**	**35.5**	**34.4**	**39.4**
Direct	22.9	22.4	22.0	24.8	23.5	23.3	27.1	27.4	31.9	28.3	32.0	32.7	29.9	27.9	29.2	31.4
Volatilization	1.1	1.0	1.1	1.2	1.1	1.2	1.2	1.3	1.4	1.4	1.5	1.4	1.3	1.5	1.3	1.4
Leaching & Runoff	5.4	5.0	4.6	4.4	3.6	4.1	5.2	4.1	6.0	3.6	5.6	6.7	5.1	6.1	3.9	6.5
Wheat	**27.8**	**24.6**	**26.1**	**34.1**	**23.8**	**25.1**	**30.6**	**27.5**	**24.6**	**19.8**	**21.7**	**20.1**	**19.6**	**19.3**	**19.2**	**19.9**
Direct	24.9	21.6	23.2	24.8	21.0	19.4	26.2	22.9	21.2	17.6	19.6	18.3	18.2	17.7	18.0	18.1
Volatilization	0.8	0.7	0.7	0.7	0.7	0.6	0.7	0.6	0.6	0.6	0.6	0.5	0.6	0.6	0.6	0.5
Leaching & Runoff	2.1	2.3	2.3	8.7	2.1	5.1	3.7	4.0	2.8	1.7	1.5	1.2	0.8	1.0	0.6	1.2
Hay	**8.6**	**7.9**	**8.2**	**4.4**	**7.9**	**8.1**	**8.9**	**7.7**	**8.9**	**8.5**	**4.5**	**9.1**	**8.4**	**7.9**	**7.7**	**8.3**
Direct	6.3	5.9	6.2	3.1	6.3	5.9	6.8	5.9	6.8	6.3	3.3	6.9	6.5	6.1	6.1	6.4
Volatilization	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4
Leaching & Runoff	1.9	1.6	1.6	1.0	1.3	1.8	1.7	1.4	1.7	1.8	0.7	1.8	1.4	1.5	1.2	1.4
Cotton	**5.6**	**6.5**	**6.0**	**6.4**	**6.3**	**7.3**	**6.8**	**6.4**	**6.5**	**6.8**	**7.8**	**7.5**	**6.9**	**5.4**	**5.7**	**6.3**
Direct	4.8	5.0	5.2	5.4	5.2	5.9	5.7	5.3	5.6	5.7	6.4	6.4	5.3	4.6	4.7	5.0
Volatilization	0.1	0.2	0.2	0.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.2
Leaching & Runoff	0.7	1.3	0.7	0.8	0.9	1.3	0.9	1.0	0.8	0.9	1.2	0.9	1.4	0.7	0.8	1.1
Sorghum	**3.5**	**4.2**	**4.9**	**5.0**	**2.9**	**4.1**	**4.2**	**4.8**	**4.1**	**4.7**	**3.4**	**5.3**	**3.4**	**4.1**	**2.5**	**3.1**
Direct	2.9	3.9	3.8	4.5	2.6	3.8	3.9	4.2	3.5	3.9	2.9	5.0	3.1	3.8	2.3	2.9
Volatilization	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0
Leaching & Runoff	0.6	0.2	1.0	0.5	0.2	0.3	0.2	0.5	0.5	0.7	0.4	0.3	0.2	0.2	0.1	0.2
Non-major crops	**19.2**	**17.2**	**15.7**	**18.8**	**22.8**	**21.1**	**20.3**	**21.4**	**21.5**	**23.7**	**21.5**	**19.0**	**18.1**	**22.4**	**22.7**	**23.6**
Direct	14.4	15.2	14.2	16.0	19.9	18.1	17.8	18.6	18.6	20.4	18.8	16.4	15.4	18.4	18.3	19.4
Volatilization	1.6	0.5	0.3	0.7	0.8	0.8	0.7	0.8	0.8	0.9	0.7	0.7	0.7	1.1	1.3	1.2
Leaching & Runoff	3.2	1.5	1.2	2.0	2.1	2.2	1.9	2.1	2.2	2.4	2.0	1.9	2.0	2.9	3.2	3.0
Histosol Cultivation[2]	**2.8**	**2.8**	**2.8**	**2.8**	**2.8**	**2.8**	**2.8**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**	**2.9**
Managed Manure[3]	**7.5**	**7.7**	**7.7**	**7.9**	**7.9**	**8.0**	**7.9**	**8.2**	**8.2**	**8.3**	**8.4**	**8.5**	**8.7**	**8.5**	**8.5**	**8.5**
All Direct	130.5	130.1	129.8	138.2	136.8	128.9	149.7	145.4	152.0	141.9	146.2	150.5	140.0	132.3	137.5	142.3
All Volatilization	5.9	4.4	4.6	4.7	4.9	4.9	4.9	5.1	5.3	5.2	5.3	5.0	5.0	5.4	5.4	5.5
All Leaching & Runoff	24.6	19.7	19.3	26.0	15.1	21.8	21.8	18.3	22.8	17.5	18.8	21.1	17.3	20.2	14.8	20.7
Total	**161.0**	**154.3**	**153.7**	**168.9**	**156.8**	**155.6**	**176.4**	**168.7**	**180.0**	**164.6**	**170.4**	**176.6**	**162.3**	**158.0**	**157.7**	**168.4**

[1] Emissions from residue burning are not included.

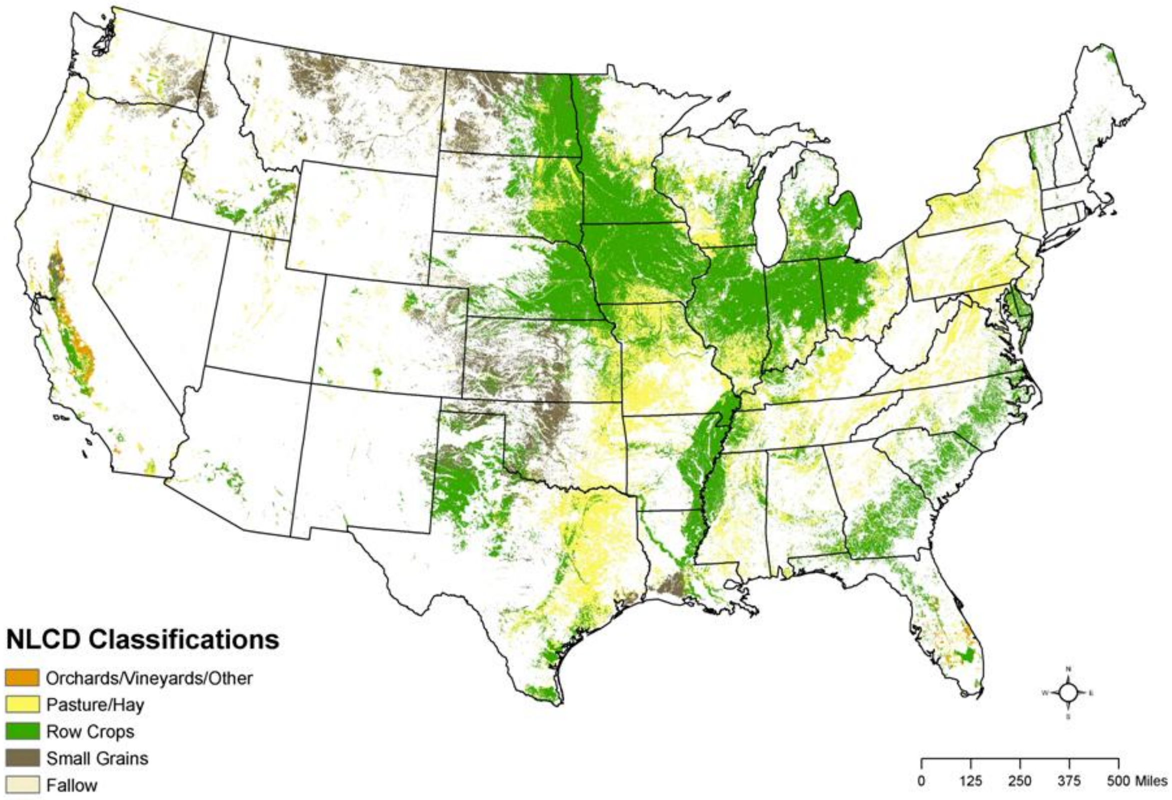
[2] Direct emissions.

[3] Accounts for loss of manure N during transport, treatment and storage, including both volatilization and leaching/runoff.

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. In contrast to the first edition of the USDA GHG report (USDA 2004) that relied exclusively on IPCC (1997) methodology, this edition includes estimates for N_2O emissions and CO_2 fluxes from cropped soils obtained from the DAYCENT and CENTURY ecosystem models. Another change compared to the 1st edition is that CO_2 fluxes for grazed lands that were previously included in this chapter are now included in the Livestock and Grazed Land chapter.

Map 3-2

U.S. Cropped Land



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 and 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 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 0.5 to 5 % in the upper 20-30 cm and so they are classified as mineral soils. 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). 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 20 to 30 % organic matter by weight, and constitute a special case. 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 universally on flooded fields, including in the U.S. where rice is grown exclusively on flooded fields (EPA 2007). 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

Crop residues are sometimes burned in fields to prepare for cultivation and control for pests, although this is not a common practice in the U.S. (EPA 2007). 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. (EPA 2007).

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 and 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

On average, ~85% of total cropland soil N_2O emissions are direct soil emissions (Table 3-3). Of the 15% of total emissions from indirect N_2O, 80% are from NO_3 leaching/runoff and the remainder are associated with volatilization. Corn cropland has the highest emissions, almost 40% of the total, followed by soybean and wheat (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. Although wheat area has tended to decline, it still covers an area comparable to soybean and is the third highest in emissions. Emissions from hay cropping are also substantial. Emissions from hay are lower than those from wheat even though the areas are similar because hay is not typically fertilized with N and a large portion of the N supplied by fixation by legumes (e.g., alfalfa) is removed during harvest. Emissions from cotton and sorghum are low as the cropland areas for these crops is small compared to the other major crops simulated by DAYCENT. Non-major crop types were responsible for ~14% of total emissions on average. 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 to cropped soils varied between ~14 and 17 Tg N between 1990 and 2005 (Fig. 3-2) while N_2O emissions varied between 154 and 180 Tg CO_2 eq. (Table 3-3). However, variation in N inputs only explained about 38% 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.

3.3.1 Changes Compared to the 1st edition of the USDA GHG Report

In contrast to the first edition of the USDA GHG report, this edition uses the process-based model DACENT to estimate N_2O emissions from the majority of agricultural soils in the U.S. DAYCENT simulates major crops (corn, soybean, wheat, hay, sorghum, and cotton) at county level resolution. The model simulates corn and sorghum harvested for grain and silage, and alfalfa hay as well as non-alfalfa hay. The DAYCENT simulations accounted for ~90% of synthetic nitrogen fertilizer applied to cropland soils in the U.S. and ~86% of cropland area in the U.S. IPCC (2006) emissions factor methodology was used to estimate emissions from crops not accounted for by DAYCENT (e.g., oats, tobacco, sugarcane, orchards, cash crops) and emissions associated with cultivation of histosols. IPCC (2006) methodology assumes that N_2O emissions are solely a function of N inputs to the soil. The major advantage of using DAYCENT to compute emissions is that the model accounts for additional factors that influence emissions like weather, soil type, and previous land use history, making estimates more reliable. Comparisons of observed N_2O emissions from experimental plots throughout North America with emissions estimated using DAYCENT and IPCC methodologies showed that DAYCENT was closer to the observed values (Del Grosso et al. 2005).

Another change is due to the nature of how nitrogen cycling is represented in this process-based model. Emissions cannot be partitioned as they were in the first edition. In the first edition, emissions were partitioned based on the source of nitrogen inputs (synthetic fertilizer, fixed N, crop residue, manure, etc.) because the IPCC (1997) methodology was based on N inputs. With DAYCENT, once nitrogen enters the plant/soil system, it can be taken up by vegetation, metabolized by microbes, or stored in the soil, and also cycled among these components. Consequently, when the model simulates emission of a given amount of nitrogen gas, it is impossible to accurately distinguish the original source of the nitrogen. Instead of partitioning N_2O emissions by nitrogen input type, emissions are partitioned spatially and by crop type.

Another major change in this edition compared to the first relates to prior assumptions about synthetic nitrogen fertilizer. Instead of assuming that all of the synthetic nitrogen fertilizer sold in this country was applied to agricultural soils, this edition accounts for the portion of total fertilizer that was applied for non-farm use (e.g., golf courses, parks, lawns) based on data compiled by the USGS (Ruddy et al. 2006). The following sections present emission estimates obtained by summing DAYCENT estimates for the major crop listed above and IPCC (2006) estimates for other crops. Following this, the methodologies used to conduct the DAYCENT simulations for major crops and IPCC methodology for other crops are summarized. Lastly, a quantification of N_2O mitigation is included in this edition.

3.3.2 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 2007), with some exceptions. The U.S. GHG Inventory includes in Agricultural Soils Management direct emissions of N_2O from livestock on grazed lands, while the USDA GHG Inventory includes this source under Livestock GHG Emissions. The methodology outlined below does not include the portion of N_2O emissions from grazed lands. Methods for this source are covered in Chapter 2 of this report. Also, the U.S. GHG Inventory includes in Agricultural Soils Management 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 (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 two 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. 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 0th 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 2003). 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 1600 years of native vegetation was needed to initialize soil organic matter (SOM) pools in the model. Modern weather (1980-2003) 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 1990-2005.

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 predominates.

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; NASS 2004, 1999, 1992; Grant and 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 (2007) provided supplemental data on county-level variation in manure production across the time series from 1990 to 2005. 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. 2006) 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 0th 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 and 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 0th simulation for each crop in each county in each agricultural region were adjusted by comparing the 0th 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, 1990). DAYMET weather data is available for the United States at 1 km^2 resolution for 1980 through 2003.

Soil Properties: Soil texture data required by DAYCENT was obtained from STATSGO (Soil Survey Staff, Natural Resources Conservation Service, 2005), and was 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 (NASS 1992, 1999, 2004). 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 manure N application rates between 1990 and 2005 were obtained by multiplying the amount of manure N produced in that year by the proportion of manure N applied to the same crop in 1997; the amount of land receiving manure (approximately 5 percent of total cropped land) was assumed to be constant during 1990 through 2005. 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).

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[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.

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 2005 (USDA 2005b), and this 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 above and below ground 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).

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 (1994, 1998, 2000a, 2001, 2002, 2003), Schueneman (1999a, 1999b, 1999c, 2001), Deren (2002), Schueneman and Deren (2002), Cantens (2004), Lee (2003, 2004).

- Crop residue N was derived by combining amounts of above- and below-ground biomass, which were determined based on crop production yield statistics (USDA 1994, 1998, 2003, 2005b, 2006b), 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 1991, 1992a, 1993, 1994; AAPFCO 1995, 1996, 1997, 1998, 1999, 2000a, 2000b, 2002, 2003, 2004, 2005, 2006). 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 onfarm 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 National Resources Inventory (USDA 2000b, as extracted by Eve 2001 and amended by Ogle 2002), using temperature and precipitation data from Daly et al. (1994, 1998). 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 2000), 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 2000).

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; 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. Model structure uncertainty was quantified by comparing DAYCENT estimates of N_2O emissions with measured values. Uncertainty for direct emissions from minor crops and indirect emissions from all crops were estimated using simple error propagation (IPCC 2006). 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 137 and 188 Tg CO_2 eq. (Table 3-1).

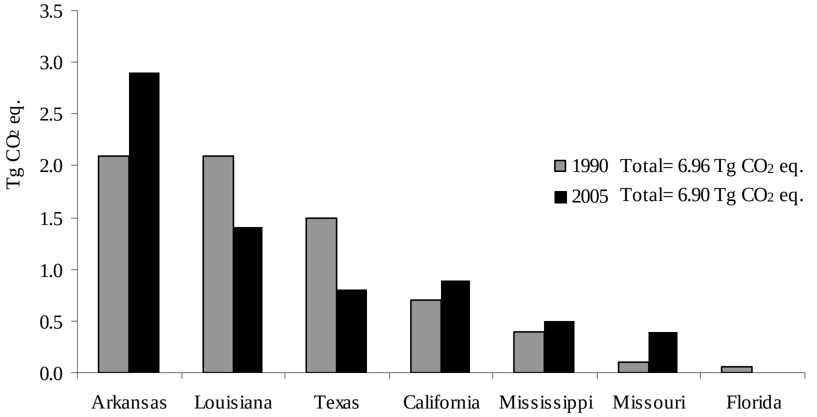
3.3.5 Mitigation of N_2O Emissions

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 (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 are decreased but also lower NO_3 leaching rates because reduced nitrification also reduces inputs to the soil NO_3 pool. Unfortunately, as with time-released fertilizer, fertilizer amended with nitrification inhibitors is more expensive. Further analyses of the environmental and economic costs and benefits of the different mitigation strategies needs to be performed before optimum mitigation strategies can be identified.

3.4 Methane Emissions from Rice Cultivation

Figure 3-3

Methane from Rice Cultivation by State, 1990 & 2005



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 2007). Methane emissions from primary and ratoon crops are accounted for separately because emissions are higher from ratoon crops (EPA 2007). Overall, rice cultivation is a small source of CH_4 in the United States. In 2005, CH_4 emissions totaled 6.9 Tg CO_2 eq, of which 6.0 Tg CO_2 eq. were from primary crops in all seven States and 0.9 Tg CO_2 was from ratoon crops in four States (Table 3-4).

Arkansas and Louisiana had the highest CH_4 emissions from rice cultivation in 2005, followed by California and Texas. Missouri and Florida both had emissions of less than 0.5 Tg CO_2 eq. (Table 3-4). Overall since 1990, CH_4 emissions from rice cultivation have decreased almost 3% (Table 3-5). While small national-scale changes were seen between 1990 and 2005 (3% decrease), sizeable shifts occurred at State levels during that time period. For example, CH_4 emission in Missouri, Arkansas and California increased by 180%, 35% and 28%, respectively, while emissions in Florida declined by 68% (Table 3-5). Although CH_4 emissions from Missouri increased by 180% between 1990 and 2005, 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 1990-2005.

3.4.1 Methods for Estimating CH_4 Emissions from Rice Cultivation

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 2007). 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

Table 3-4 Methane from Rice Cultivation from Primary and Ratoon Operations by State, 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
	Tg CO_2 eq.															
Primary	**5.1**	**5.0**	**5.6**	**5.1**	**6.0**	**5.6**	**5.0**	**5.6**	**5.8**	**6.3**	**5.5**	**5.9**	**5.7**	**5.4**	**6.0**	**6.0**
Arkansas	2.1	2.2	2.5	2.2	2.5	2.4	2.1	2.5	2.7	2.9	2.5	2.9	2.7	2.6	2.8	2.9
California	0.7	0.6	0.7	0.8	0.9	0.8	0.9	0.9	0.8	0.9	1.0	0.8	0.9	0.9	1.1	0.9
Florida	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Louisiana	1.0	0.9	1.1	0.9	1.1	1.0	1.0	1.0	1.1	1.1	0.9	1.0	1.0	0.8	1.0	0.9
Mississippi	0.4	0.4	0.5	0.4	0.6	0.5	0.4	0.4	0.5	0.6	0.4	0.5	0.5	0.4	0.4	0.5
Missouri	0.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.3	0.3	0.4	0.3	0.3	0.3	0.4
Texas	0.6	0.6	0.6	0.5	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.3	0.4	0.4
Ratoon	**2.1**	**2.0**	**2.2**	**1.9**	**2.3**	**2.1**	**1.9**	**1.9**	**2.1**	**2.0**	**2.0**	**1.7**	**1.1**	**1.5**	**1.6**	**0.9**
Arkansas	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Florida	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.1	+	+	+
Louisiana	1.1	1.0	1.2	1.1	1.2	1.1	1.1	1.2	1.2	1.2	1.3	1.1	0.5	1.0	1.1	0.5
Texas	0.9	0.9	0.9	0.8	0.9	0.8	0.8	0.7	0.8	0.7	0.7	0.6	0.5	0.5	0.6	0.4
Total	**7.1**	**7.0**	**7.9**	**7.0**	**8.2**	**7.6**	**7.0**	**7.5**	**7.9**	**8.3**	**7.5**	**7.6**	**6.8**	**6.9**	**7.6**	**6.9**

(+) Less than 0.05 Tg CO_2 Eq.

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

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.

Table 3-5 Change In Methane Emissions from Rice Cultivation, 1990-2005

	1990	2005	% Change 1990-2005
State	*Tg CO_2 eq.*		
Arkansas	2.14	2.90	35%
California	0.70	0.90	28%
Florida	0.06	0.02	-68%
Louisiana	2.06	1.40	-32%
Mississippi	0.45	0.50	12%
Missouri	0.14	0.40	180%
Texas	1.57	0.80	-49%
Total	**7.12**	**6.92**	**-3%**

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 and Bollich 1993, Lindau et al. 1995) 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 2001 for all States except Florida were taken from USDA NASS Field Crops Final Estimates 1987-1992 (USDA 1994), Field Crops Final Estimates 1992-1997 (USDA 1998), Crop Production 2000 Summary (USDA 2001), and Crop Production 2001 Summary (USDA 2002). Harvested rice areas in Florida, which are not reported by USDA, were obtained from Tom Schueneman (1999b, 1999c, 2000, 2001), a Florida agricultural extension agent, and Dr. 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.

In Arkansas, ratooning occurred only in 1998 and 1999, when the ratoon area was less than 1% of the primary area (Slaton 1999, 2000, 2001). 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 2001a), 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 1999a, 2001, 2002 and 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 1999a,1999b, 2000, 2001, 2002).

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 2007). The information is reproduced here with permissions from the EPA.

Methane emissions 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, from variation in cultivation practices, fertilizer application, 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 is 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 less than 10% 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. Estimates of the area of off-season flooding range from five to 68% of the rice acreage. Fields are flooded for a variety of reasons: to provide habitat for waterfowl, to provide ponds for crawfish production, and to aid in rice straw decomposition.

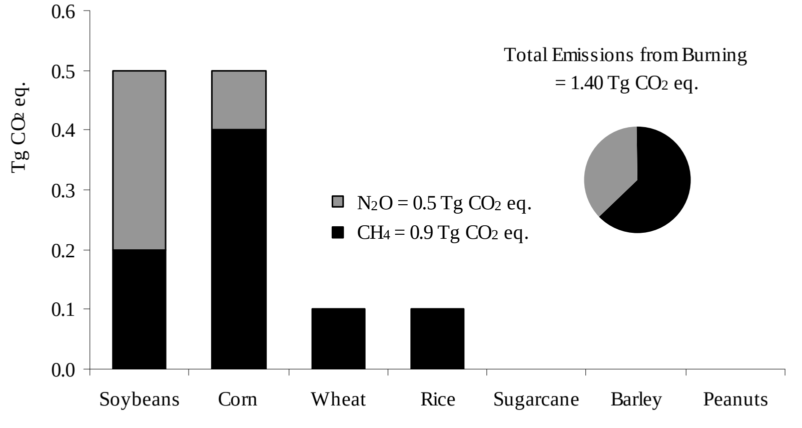
A Monte Carlo analysis was performed to quantify the uncertainties mentioned above. The calculated 95% confidence interval was 2.1 to 18.6 Tg CO^2 eq. for CH_4 emissions from rice cultivation, or 70% below and 170% above the estimate of 6.9 Tg CO^2 eq. (Table 3-1).

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 2007). For most crops, residues are burned per year, but a higher portion of rice residues is burned annually (EPA 2007). Consequently, emissions from residue burning are a small source of overall crop-related emissions in the U.S.

Figure 3-4

Greenhouse Gas Emissions from Field Burning by Crop Type, 2005

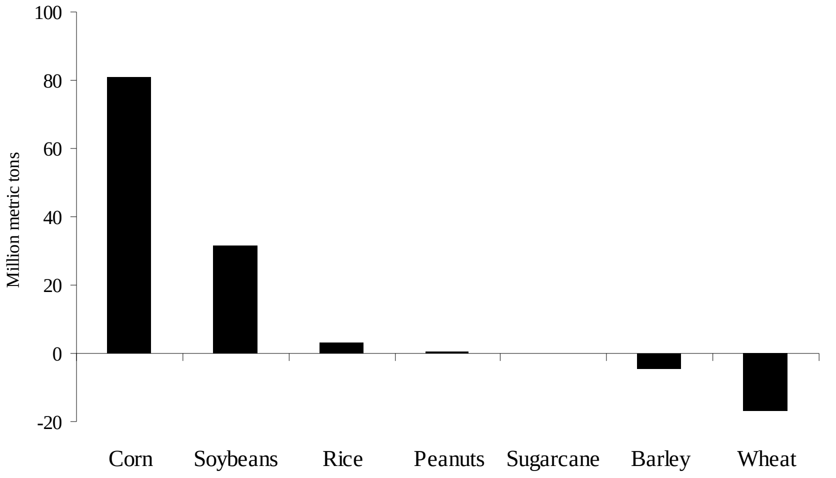


About three-fifths of GHG emissions from residue burning, across all crop types, consisted of CH_4 in 2005; the remaining was N_2O (Table 3-6, Figure 3-4). The highest GHG emissions were from burning of soybean and corn crop residues, at 40% each. Burning of wheat, rice, sugarcane, and barely crop residues each contributed 10% 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.

Total GHG emissions from residue burning increased 33% from 1990 to 2005. Trends in relative GHG emissions were similar across crop types in 1990 compared to 2005 with a few exceptions. In 1990, burning of corn residues contributed the most to GHG emissions from residue burning, while burning of soybeans was the second largest source. By 2005, these crops had similar emissions form burning. Between 1990 and 2005, soybean and corn production both increased in absolute amounts (Figure 3-5). However, proportionally, soybean production increased more dramatically than corn (soybean production increased by 62% and corn by 50%) (Figure 3-6). In addition, soybeans have higher nitrogen content than corn, resulting in greater N_2O emission per unit of crop mass burned. Thus, while corn production was still greater than soybean production in 2005, GHG emissions from soybean residue burning were about equal to those from corn residue burning.

Figure 3-5

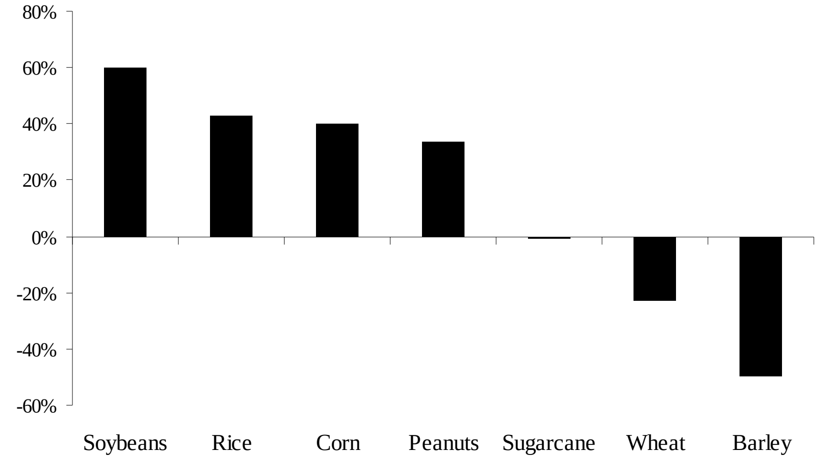
Change in Commodity Production, 1990-2005



Appendix Table B-2 provides the complete time series of crop production from 1990 to 2005 for crop types that contribute to GHG emissions from burning, Appendix Table B-3 provides crop production by State of crops managed with burning for 2005.

Figure 3-6

Percent Change in Commodity Production, 1990-2005



Illinois and Iowa had the highest State levels of GHG emissions from residue burning in 2005, emitting roughly 0.15 and 0.19 Tg CO_2 eq., respectively, of CH_4 and N_2O combined (Appendix

Table 3-6 Greenhouse Gas Emissions from Agriculture Burning by Crop, 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Source	*Tg CO_2 eq.*															
Methane	**0.7**	**0.6**	**0.8**	**0.6**	**0.8**	**0.7**	**0.7**	**0.8**	**0.8**	**0.8**	**0.8**	**0.8**	**0.7**	**0.8**	**0.9**	**0.9**
Wheat	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Rice	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Sugarcane	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corn	0.3	0.3	0.3	0.2	0.4	0.3	0.3	0.3	0.4	0.3	0.4	0.3	0.3	0.4	0.4	0.4
Barley	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Soybeans	0.1	0.2	0.2	0.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.2
Peanuts	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitrous oxide	**0.4**	**0.4**	**0.4**	**0.3**	**0.5**	**0.4**	**0.4**	**0.4**	**0.5**	**0.4**	**0.5**	**0.5**	**0.4**	**0.4**	**0.5**	**0.5**
Wheat	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Rice	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sugarcane	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corn	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Barley	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Soybeans	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.3	0.3
Peanuts	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total	**1.1**	**1.0**	**1.2**	**0.9**	**1.3**	**1.0**	**1.2**	**1.2**	**1.2**	**1.2**	**1.2**	**1.2**	**1.1**	**1.2**	**1.4**	**1.4**

Table B-5 and Appendix Table B-6). The next highest levels of GHG emissions from residue burning were in order Iowa, Illinois, Minnesota, Nebraska, Indiana, Arkansas, Kansas, and Ohio, with emissions between 0.06 and 0.11 Tg CO_2 eq. State-level GHG emissions from residue burning are strongly tied to crop production. State-level estimates of crop production are provided in Appendix Table B-3 for corn, soybeans, wheat, rice, sugarcane, barley, and peanuts.

3.5.1 Methods for Estimating CH_4 and N_2O Emissions from Residue Burning

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 2007). 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: kl
The crop residues that are burned in the United States were determined from various State-level GHG emission inventories (ILENR 1993, Oregon Department of Energy 1995, Wisconsin Department of Natural Resources 1993) and publications on agricultural burning in the United States (Jenkins et al. 1992, Turn et al. 1997, EPA 1992). Crop production data for these crops, except rice in Florida, were taken from USDA's *Field Crops Final Estimates* 1987-1992, 1992-1997, 1997-2002 (USDA 1994, 1998, 2003b) and *Crop Production 2004 Summary* (USDA 2005a). Rice production data for Florida were estimated by applying average primary and ratoon crop yields for Florida (Schueneman and Deren 2002) to Florida acreages (Schueneman 1999b, 2001; Deren 2002; Kirstein 2003, 2004; Cantens 2004, 2005).

Residue-to-Crop Product Mass Ratios:
All residue/crop product mass ratios except sugarcane were obtained from Strehler and Stützle (1987) and Meisinger and Randall (1991). The ratio for sugarcane is from the University of California (1977).

Fraction of Residues Burned:
The percentage of crop residue burned was assumed to be three percent for all crops in all years, except rice, based on State inventory data (ILENR 1993, Oregon Department of Energy 1995, Noller 1996, Wisconsin Department of Natural Resources 1993, Cibrowski 1996). Estimates of the percentage of rice acreage on which residue burning took place were obtained on a State-by-State basis from agricultural extension agents in each of the seven rice-producing States (Bollich 2000; Deren 2002; Guethle 1999, 2000, 2001, 2002; Fife 1999; California Air Resources Board 1999; Klosterboer 1999a, 1999b, 2000, 2001, 2002; Linscombe 1999a, 1999b, 2001, 2002; Mutters 2002, Najita 2000, 2001; Schueneman 1999a, 1999b, 2001; Slaton 1999a, 1999b, 2000; Street 1999, 2000, 2001, 2002; Wilson 2004, 2005) (Appendix B-4).

The estimates provided for Florida remained constant over the entire 1990-2005 period, while the estimates for all other States varied over the time series. For California, it was assumed that the annual percent of rice acreage burned in Sacramento Valley is representative of burning in the entire State, because the Sacramento Valley accounts for over 95% of the rice acreage in California (Fife 1999). The annual percent of rice acreage burned in the Sacramento Valley was obtained from staff at the California Air Resources Board (CARB) (Najita, 2001), a report of the CARB (2001), and background data for future editions of the report (Lindberg 2002). These values declined over the period 1990 through 2005 because of a legislated reduction in rice straw burning.

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 Efficiency:
Burning efficiency refers to the fraction of dry biomass exposed to burning that actually burns. The burning efficiency was assumed to be 93%.

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). The nitrogen content of peanuts is from Ketzis (1999).

Combustion Efficiency:
Combustion efficiency refers to the fraction of carbon in the fire that is oxidized completely to CO_2. Combustion efficiency was assumed to be 88% for all crop types (EPA 1994).

State-level emissions estimates were calculated with the above equations, applying State-level production data to national-level coefficients. The State-level rice estimates were provided directly by EPA, using State-specific residue fractions (the fraction of residues burned varies among States), and State production data.

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 2007). 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 0.45 to 57 Tg CO^2 eq. for N_2O emissions from residue burning, or 10% below and 14% above the estimate of 0.5 Tg CO^2 eq. and 0.75 to 0.97 Tg CO^2 eq. for CH_4 emissions from residue burning, or 17% below and 8% above the estimate of 0.9 Tg CO^2 eq. (Table 3-1).

3.6 Carbon Stock Changes in Cropped Soils

In contrast to the first edition of the USDA GHG report, this edition uses the process-based model CENTURY to estimate CO_2 fluxes from the majority of agricultural soils in the U.S. CENTURY simulates most crops except vegetables, tobacco, horticultural crops, orchards, rice, and crops grown on organic soils. An IPCC (2006) Tier 2 approach was used to estimate fluxes from all crops not simulated by CENTURY. The IPCC (2006) methodology calculates soil C changes based on previous and current land use. The major advantage of using CENTURY to estimate soil C changes is that the model accounts for additional factors that influence C levels like weather, soil type, and fertilizer additions, making estimates more reliable.

3.6.1 Emissions by Land Use

Except for cultivated organic soils and liming practices, cropped soils in the U.S. were estimated to accumulate about 66.5 Tg CO_2 eq. in 2005 (Table 3-1)[4]. Much of the carbon change is 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 and pasture also impact the estimated 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 (1990-2005). About 30 Tg CO_2 eq. was emitted from cultivation of these soils in 2005 (Table 3-1). Liming of agricultural soils resulted in emissions of about 4 Tg CO_2 eq per year. Total net carbon sequestration in 2005 was about 32 Tg CO_2 eq. when all of the above components were taken into consideration. Carbon uptake on agricultural soils varied between 1990 and 2005 (Table 3-

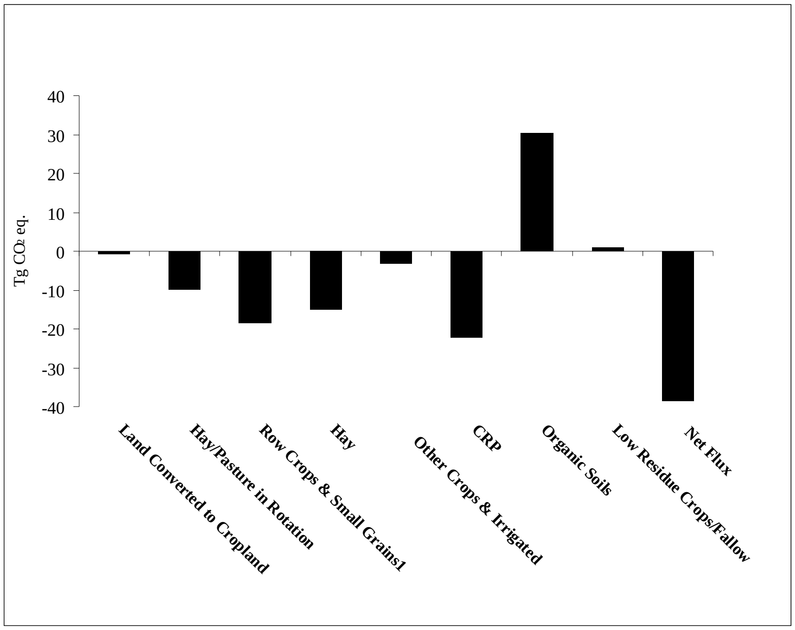


Figure 3-7. CO_2 Emissions and Sequestration from Cropland Soils, 2005

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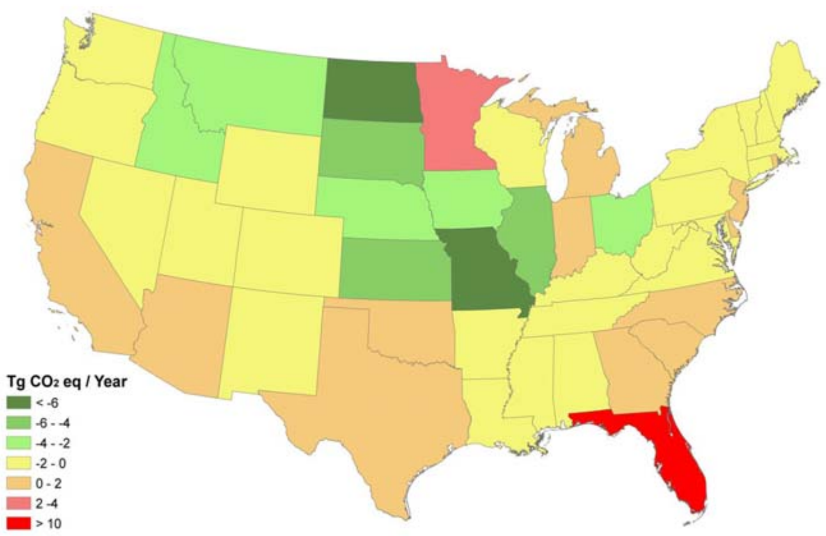
[4] Emissions and sinks of carbon in agricultural soils are expressed in terms of CO_2 equivalents; carbon sequestration is a result of changes in stocks of carbon in soils, from which CO_2 fluxes are inferred. Units of CO_2 equivalent can be converted to carbon using a multiplier of 0.272.

2), driven largely by land use changes and weather fluctuations.



Map 3-3
State Level Carbon Dioxide Fluxes from Cropped Soils in 2005

Most States in the Corn Belt are storing C in cropped soils due to adoption of reduced tillage 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.

3.6.2 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 Gas Inventory (EPA 2007).

3.6.2.1 CENTURY Model Simulations for Most Cropped Mineral Soils

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, Ross 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. MLRA's represent geographical units with relatively similar soils, climate, water resources, and land use. Data on historical cropping practices for different regions were obtained form 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 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 data base. 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 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. 2003; 2006b). Carbon stocks are listed in Appendix Table B-7, stock change rates are listed in Appendix Table B-8, areas of cropped organic soils are listed in Appendix Table B-9, and carbon loss rates from organic soils are listed in Appendix Table B-10.

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.

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. Land areas grouped by major land use and management system types are shown in Appendix Table B-11, carbon stock changes by State and land use/management in Appendix Table B-12, and by State on cropland by major activity in Appendix Table B-13. Tillage practices are not included in the NRI. Thus, supplemental data were used from the Conservation Technology Information Center (CTIC 1998),

Table 3-7 Tillage Percentages by Management Category and Climate Zones[1]

	1982			**1992**			**1997**		
Climate & System	No Till[2]	Red. Till[3]	Conv. Till[4]	No Till	Red. Till	Conv. Till	No Till	Red. Till	Conv. Till
STD									
Continuous Cropping Rotations[5]	0	3	97	0	4	96	0	15	85
Rotations with Fallow[6]	0	0	100	0	2	98	0	5	95
Low Residue Ag.[7]	0	3	97	0	4	96	0	10	90
STM									
Continuous Cropping Rotations	0	0	100	0	20	80	1	10	89
Rotations with Fallow	0	0	100	0	10	90	1	10	89
Low Residue Ag.	0	3	97	0	4	96	0	5	95
WTD									
Continuous Cropping Rotations	0	0	100	0	10	90	1	15	84
Rotations with Fallow	0	3	97	0	15	85	2	20	78
Low Residue Ag.	0	3	97	0	1	99	0	0	100
WTM									
Continuous Cropping Rotations	0	6	94	10	30	60	12	28	60
Rotations with Fallow	0	6	94	5	30	65	8	27	65
Low Residue Ag.	0	9	91	1	10	89	2	13	85
CTD									
Continuous Cropping Rotations	0	3	97	2	25	73	8	12	80
Rotations with Fallow	0	6	94	4	25	71	12	13	75
Low Residue Ag.	0	0	100	1	2	97	2	6	92
CTM									
Continuous Cropping Rotations	0	11	89	5	30	65	3	17	80
Rotations with Fallow	0	11	89	5	30	65	3	27	70
Low Residue Ag.	0	0	100	1	2	97	1	7	92

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

[1] Including Adjustments for Long-term Adoption of No-till Agriculture

[2] No-till includes CTIC survey data designated as no-tillage.

[3] Reduced-till includes CTIC survey data designated as ridge tillage, mulch tillage, and reduced tillage.

[4] Conventional till includes CTIC survey data designated as intensive tillage and conventional tillage.

[5] Medium and high input rotations (based on the IPCC categories) found in Table B-9. CTIC survey data for corn, soybeans, and sorghum were used in this category.

[6] Rotations with fallow found in Table B-9. CTIC survey data on fallow and small grain cropland were used in this category.

[7] Low input rotations found in Table 3, with the exception of rotations with fallow. CTIC survey data on cotton were used in this category; tillage rates are assumed to be the same for low residue crops and vegetables in rotation.

which reports tillage practices by major crops and county on an annual basis (Table 3-7). Data for wetland restoration under the CRP program were obtained from Euliss and Gleason (2002).

Organic soils (i.e., peat, mucks) that have been drained and converted to cropland or pasture use are subject to potentially high rates of carbon loss. Annual C losses were estimated using IPCC (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 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 and McBride 2005). The methodology summarized above is described in detail chapter 7 of the U.S. GHG Inventory (EPA 2007).

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 2007).

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 ~66 Tg CO_2 eq. in 2005 with a 95 % confidence interval of +/- 16%. 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 form 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 was 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

As estimated by Tier 2 methodology, mineral soils not simulated by CENTURY sequestered ~0.5 Tg CO_2 eq. in 2005 with a 95 % confidence interval of –830 % and +832% and organic soils emitted 30.3 Tg CO_2 eq. in 2005 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 form probability distribution functions (Ogle et al. 2003). For mineral soils, probability distribution functions were derived form 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 form 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 2005 with a 95 % confidence interval of –94 % and +96 %. 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 2005 ranged from 17 to 50 Tg CO_2 eq.around the estimate of 32.2 Tg CO_2 eq. (Table 3-1).

3.8 Mitigation of CO_2 Emissions

Currently, cropped soils in the U.S. are estimated to be storing carbon at the rate of approximately 30 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. To estimate mitigation potential for this report, the amount of land currently under different land management categories and land management changes were considered. Currently, the majority of cropped land in the U.S. is fully tilled (Table 3-7). Full tillage usually does not lead to carbon storage because tillage enhances decomposition of soil organic matter. Thus, reduction in tillage intensity provides on opportunity to store carbon. Other strategies to increase soil carbon considered here are: reduced cropping of organic soils, reduced summer fallow, 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. (Table 3-8), but are a source of about 30 Tg CO_2 per year. 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.

Table 3-8 Cropland Area by Management Practice[1]

	Area	% of Total Cropland
Current Management	*million ha*	
Full Tillage	88.3	54.3 %
Reduced Tillage	28.0	17.2%
No Till	10.7	6.6%
Summer Fallow	19.0	11.7%
Hay/Pasture in Rotation	3.3	2%
Conservation Reserve Program	12.1	7.4%
Highly Erodable Lands	21.8	13.4%
Organic Soils	0.7	0.5%

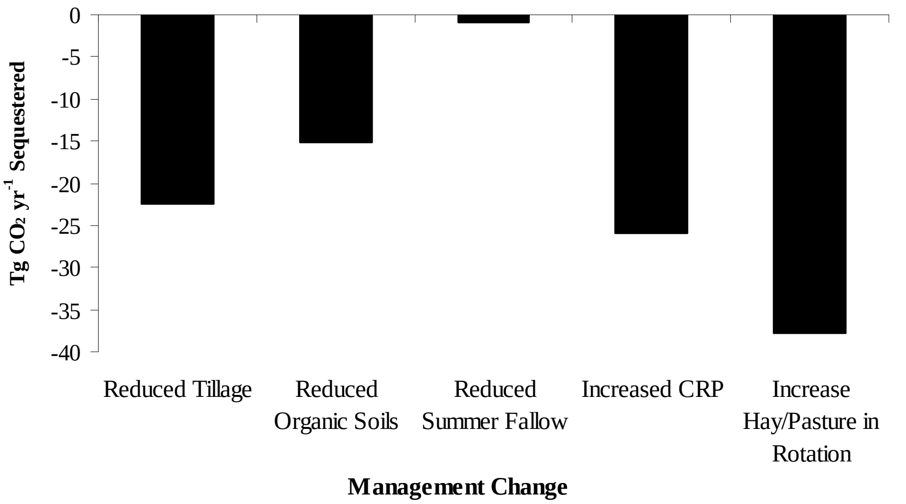
[1] Categories are not mutually exclusive, e.g., land in summer fallow is also classified by tillage intensity.

CENTURY model simulations and IPCC Tier 2 methodologies were combined to estimate soil carbon stock changes for different land uses. NRI data were used to classify current land uses (Table 3-7). To estimate mitigation potential, 50% adoption with improved land use was assumed for the mitigation options considered. That is, 50% of the land in full tillage was assumed to be converted to minimum tillage, 50% of the land in minimum tillage was assumed to be converted to no till, 50% of land with summer fallow and 50% of land cropped on organic soils were assumed to be taken out of production, 50% of highly erodible lands were assumed to be converted to CRP, and 50% of crop rotations that currently do not include hay or pasture were assumed to be modified to include one or both of these in the rotation. All of these options stored large amounts of carbon except reduced summer fallow (Figure 3-8).

Together, adoption of these options could store ~104 Tg CO_2 per year; this is in addition to the ~32 Tg CO_2 per year stored currently in cropped soils. One hundred percent adoption would store a total of almost 240 Tg CO_2 per year. However, it must be pointed out that some of these strategies would affect the flux of other greenhouse gases and have other impacts. For example, taking organic soils out of production and allowing them to revert back to wetlands would store carbon but also increase methane emissions. Also, conversion to no till can increase N_2O emissions from some soils (Six et al. 2004) and sometimes lead to lower yields (Wilhelm & Wortmann 2004; Hammel et al. 1995; Lund et al. 1993), although these trends are far from universal and measures can be taken, e.g., improved nitrogen management and strip tillage, to eliminate or minimize these negative impacts. Also, it is probably not realistic to assume that 100% adoption of some strategies, such as including hay and pasture in rotations, is feasible because the extra hay produced would not necessarily be marketable.

Figure 3-8

Future Carbon Dioxide Mitigation Potential; Adoption of Management Change by 50% of Farmers



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

4.1 Summary

Forest ecosystems and forest products represent significant carbon sinks in the United States, approximately equal to 10% of total U.S. greenhouse gas (GHG) emissions (EPA 2007). The net amount of carbon stored—that is, annual incremental increase—by forests in the conterminous U.S. increased by an estimated 595 and 103 Tg CO_2 eq. in 2005 for forest ecosystems and harvested wood products, respectively. Total Sequestration in 2005 was estimated to be 699 Tg CO_2 eq. and the calculated 95% confidence interval for this flux was -890 to -513 Tg CO_2 eq. (Table 4-1). Compared to 1990, CO_2 sequestered by forest systems was about 17% greater in 2005 (Table 4-2). Current total carbon stocks in forest ecosystems of the conterminous United States are estimated at about 150 Pg CO_2 eq. (Table 4-2, Pg=1,000 Tg).

Table 4-1 Forest Carbon Stock Change Estimates and Uncertainty Intervals for 2005

	Estimate	95% Confidence Interval
Source	*Tg CO_2 eq.*	
Forest	(595)	(785) to (410)
Harvested Wood	(103)	(130) to (79)
Total	**(699)**	**(890) to (513)**

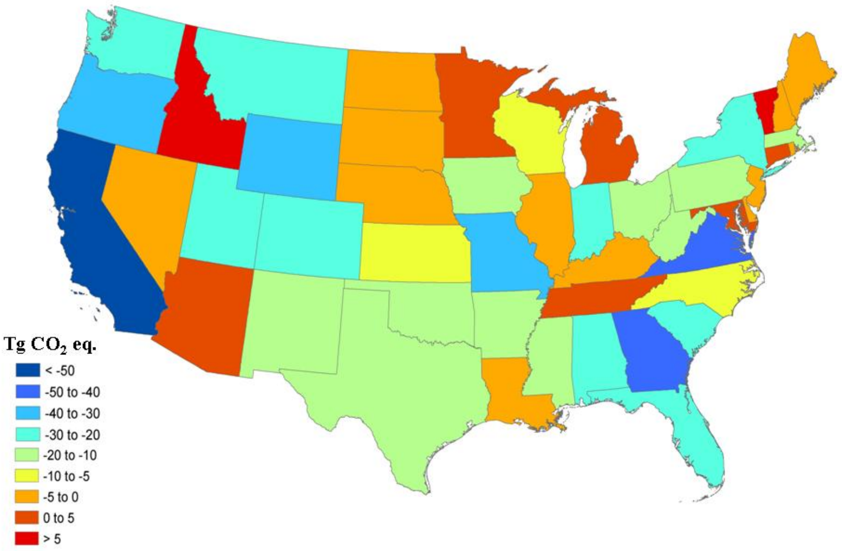
Note: Parentheses indicate net sequestration.

Periodic summary statistics on forestland in the conterminous United States indicate an approximately 2% increase in area over the interval from 1987 to 2002, that is, 246 to 251 million hectares (Smith et al. 2004a). 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 Virginia, Georgia, Oregon, Missouri, Wyoming, Montana, and Indiana (Map 4-1). Only eight States are losing forest carbon. Forest biomass, and thus carbon stocks, is greatest in The Pacific States, lowest in the Great Plains, and intermediate in the Rocky Mountain and Eastern States (Map 4-2). Eastern forests are storing slightly more carbon than Western forests (341 vs. 320 Tg CO_2 eq. yr^{-1}) but Western forests are sequestering carbon at a rate about 50% greater than Eastern forests on a per-hectare basis (3300 kg CO_2 Eq. ha^{-1} yr^{-1} vs. 2200 kg CO_2 Eq. ha^{-1} yr^{-1} in Eastern US forests, Table 4-3). Sequestration was greatest in the East in Oak/Hickory forests (246 Tg CO_2 Eq. per year). Of the Western forest types, California mixed conifers sequestered the most at 220 Tg CO_2 Eq. per year (Table 4-3).

Map 4-1

Carbon Stock Change by State in 2005



Forestlands of the United States constitute 33% (303 million hectares) of total land area. This chapter summarizes carbon stocks and stock changes on the approximately 251 million hectares located in the conterminous 48 States. This is largely because these forestlands are well-defined by inventory data – a fundamental component of these estimates and a large proportion of these forests are managed for timber production. The relative proportion subject to management is based on the 80% of the 251 million hectares that are classified as timberland, meaning they meet minimum levels of productivity and are available for timber harvest. Separate effects of management or land use change, such as afforestation, increased productivity, reduced conversion to non-forest uses, lengthened rotations, and increased proportion and retention of carbon in harvested wood products, are not individually identified, but the effects are implicitly a part of the inventory and are thus reflected in carbon stocks and stock changes. Summaries of information included in this chapter represent updates of inventories and carbon estimations relative to the national forest carbon budgets reported in the first edition of the USDA Greenhouse Gas Inventory (Smith et al. 2004b).

Map 4-2

U.S. Forest Carbon Stocks in 2005

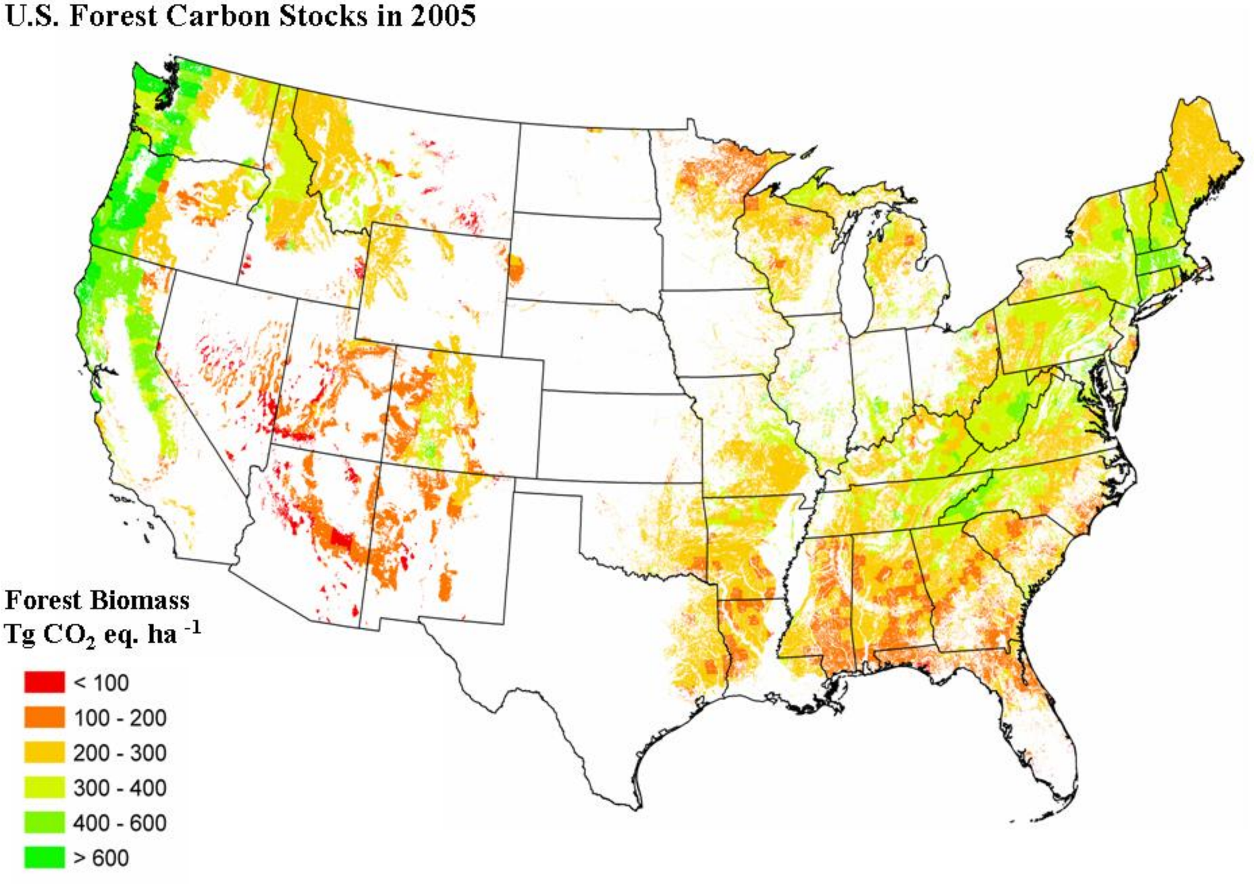


Table 4-2 Carbon Stocks and Annual Change for Forest and Wood Pools, 1990, 1998-2005[1]

	1990	1998	1999	2000	2001	2002	2003	2004	2005
Annual Change					*Tg CO_2 eq. yr^{-1}*				
Forest	**(467)**	**(584)**	**(552)**	**(529)**	**(556)**	**(595)**	**(595)**	**(595)**	**(595)**
Aboveground Biomass	(252)	(336)	(341)	(347)	(360)	(376)	(376)	(376)	(376)
Belowground Biomass	(64)	(71)	(73)	(74)	(76)	(80)	(80)	(80)	(80)
Dead Wood	(37)	(63)	(54)	(48)	(50)	(52)	(52)	(52)	(52)
Litter	(66)	(39)	(42)	(36)	(47)	(52)	(52)	(52)	(52)
Soil Organic Carbon[2]	(49)	(74)	(43)	(24)	(22)	(35)	(35)	(35)	(35)
Harvested Wood	**(132)**	**(111)**	**(116)**	**(109)**	**(90)**	**(93)**	**(91)**	**(102)**	**(103)**
Wood Products	(63)	(48)	(51)	(46)	(31)	(34)	(33)	(43)	(44)
Landfilled Wood	(69)	(63)	(65)	(63)	(59)	(59)	(58)	(59)	(59)
Total	**(599)**	**(695)**	**(668)**	**(639)**	**(646)**	**(688)**	**(687)**	**(697)**	**(699)**
Carbon Stock					*Tg CO_2 eq.*				
Forest	**143,095**	**147,644**	**148,228**	**148,780**	**149,309**	**149,865**	**150,460**	**151,055**	**151,651**
Aboveground Biomass	51,934	54,436	54,772	55,113	55,460	55,820	56,196	56,573	56,949
Belowground Biomass	10,243	10,792	10,864	10,936	11,010	11,086	11,166	11,245	11,325
Dead Wood	8,631	9,047	9,110	9,164	9,212	9,262	9,314	9,367	9,419
Litter	16,150	16,636	16,676	16,717	16,753	16,800	16,852	16,904	16,957
Soil Organic Carbon	56,138	56,733	56,807	56,850	56,875	56,896	56,931	56,966	57,001
Harvested Wood	**6,919**	**7,935**	**8,045**	**8,158**	**8,268**	**8,386**	**8,496**	**8,584**	**8,679**
Wood Products	4,341	4,814	4,866	4,917	4,965	5,016	5,064	5,093	5,130
Landfilled Wood	2,581	3,120	3,179	3,241	3,304	3,370	3,432	3,491	3,549
Total	**150,014**	**155,579**	**156,273**	**156,938**	**157,578**	**158,250**	**158,956**	**159,639**	**160,330**

Note: Parentheses indicate net sequestration

[1] Based on interpolation and extrapolation after aggregating plot-level data to state totals.

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

Estimates of stocks and net annual stock change for carbon on forestlands and in harvested wood products for the conterminous United States presented here correspond to values reported for forestlands in Chapter 7 of the U.S. GHG Inventory (EPA 2007), 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 (IPCC 2003). Thus, the forest carbon estimates reported here expand on the information provided in the U.S. GHG Inventory (EPA 2007). 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.

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	*1000 ha*		*Tg CO_2 eq.*		*Tg CO_2 eq. Yr^{-1}*		*kg CO_2/ha*
East	**155,426**	**41,248**	**11,689**	**41,596**	**(308)**	**(34)**	**(2,196)**
Aspen/Birch	7,082	1,325	434	3,413	16.0	5.3	3,005
Elm/Ash/Cottonwood	7,630	1,951	719	2,767	(20.2)	(3.3)	(3,071)
Loblolly/Shortleaf Pine	21,955	4,391	1,199	4,449	(53.5)	(10.4)	(2,914)
Longleaf/Slash Pine	5,383	827	269	1,909	(9.8)	(1.0)	(2,005)
Maple/Beech/Birch	22,416	7,229	3,099	6,909	(50.5)	(10.7)	(2,730)
Oak/Gum/Cypress	9,644	3,066	559	3,536	22.8	4.3	2,809
Oak/Hickory	54,388	16,357	3,078	9,529	(214.7)	(31.7)	(4,530)
Oak/Pine	13,114	3,025	883	2,563	7.4	3.3	815
Spruce/Fir	6,098	1,252	929	4,039	11.6	13.1	4,058
White/Red/Jack Pine	4,220	1,404	354	1,444	(5.1)	1.6	(818)
Other East Type Groups	3,497	422	165	1,038	(11.6)	(4.3)	(4,535)
West	**96,132**	**25,775**	**14,433**	**15,101**	**(242)**	**(78)**	**(3,330)**
Alder/Maple	1,390	510	151	573	n/a	n/a	n/a
Aspen/Birch	3,175	743	460	670	n/a	n/a	n/a
California Mixed Conifer	3,763	1,946	915	687	(153.0)	(66.5)	(58,338)
Douglas-fir	15,584	7,033	3,237	3,721	(54.0)	2.5	(3,304)
Fir/Spruce/Mt. Hemlock	12,345	4,931	2,886	2,020	9.8	4.4	1,153
Hemlock/Sitka Spruce	2,207	1,520	617	844	(2.4)	1.5	(375)
Lodgepole Pine	6,306	1,511	826	838	14.0	6.7	3,280
Other Western Hardwoods	1,908	270	222	241	n/a	n/a	n/a
Other Western Softwoods	1,515	352	254	194	(4.3)	(2.6)	(4,547)
Pinyon/Juniper	22,123	2,145	1,791	1,748	(24.1)	(30.3)	(2,459)
Ponderosa Pine	8,939	1,930	1,073	1,191	(20.8)	9.4	(1,279)
Redwood	261	231	98	51	(3.3)	0.2	(11,683)
Tanoak/Laurel	1,101	585	168	219	n/a	n/a	n/a
Western Larch	749	210	142	105	(7.4)	(4.4)	(15,742)
Western Oak	7,084	1,494	860	891	n/a	n/a	n/a
Western White Pine	151	42	26	25	3.5	0.8	28,724
Other West Type Groups	7,532	323	707	1,083	n/a	n/a	n/a
Total	**251,558**	**67,023**	**26,122**	**56,697**	**(549)**	**(112)**	**(2,629)**

[1] As determined from the two most recent inventories for all forests. Stock change does not include soil carbon changes.

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

Note: Parentheses indicate net sequestration.

(n/a) Indicates not available.

4.2 Concepts and Conventions

For reporting purposes, carbon estimates in forest ecosystems are allocated to the following pools (IPCC 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).

The sign convention is to assign negative net change (or flux) to carbon accumulation within forests or harvested wood pools, which we have represented here by placing numbers representing sequestration in parentheses.

Continuous, regular annual surveys are not available over the 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. (2007). 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 and annualized average information between inventory years is presented here.

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

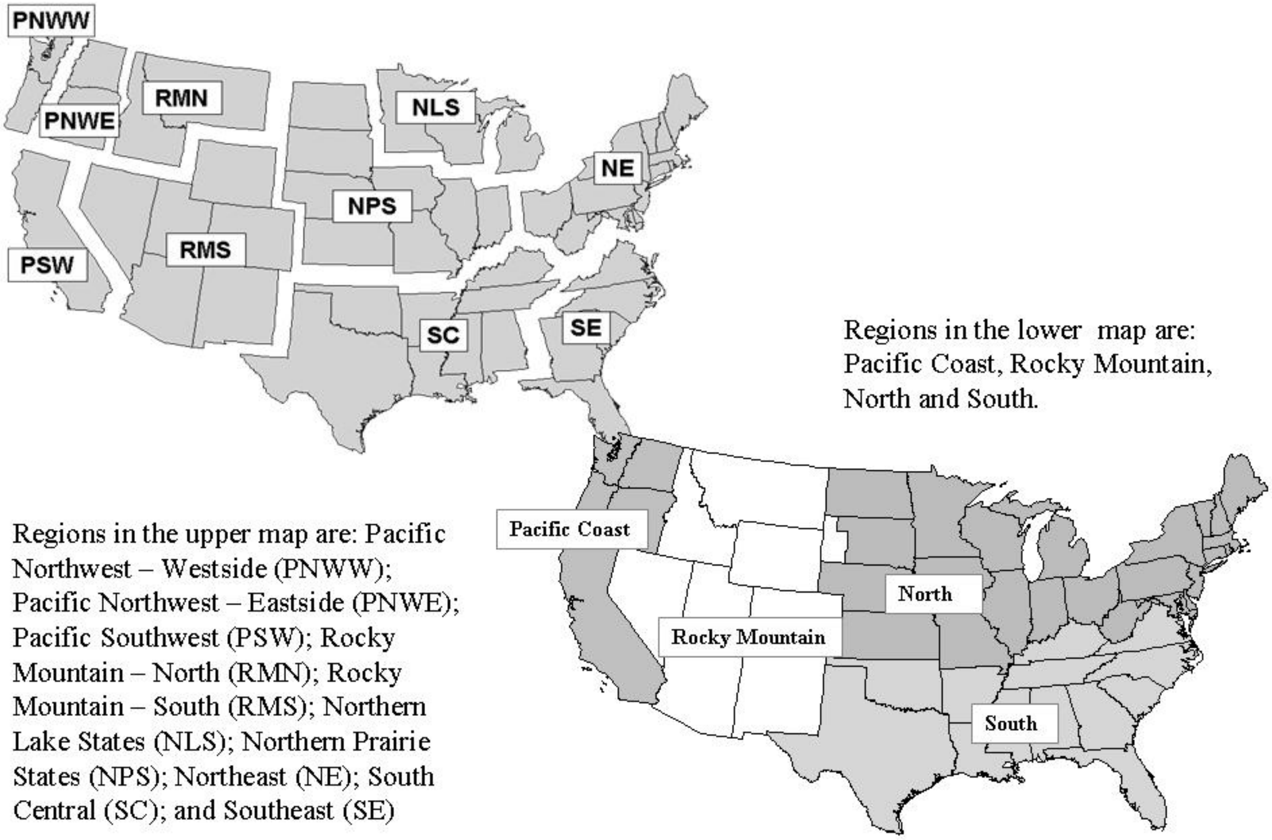
Total forest ecosystem areas, carbon stocks, and net annual stock change according to forest type group are listed in Table 4-3. Minor type groups in the East and West are pooled, for example, tropical and exotic hardwood groups in both regions. Carbon classifications in this table are for biomass, nonliving 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 2006) before calculating annualized stock or stock change. Note that changes in classifications have occurred in forest type groups definitions between the RPA and FIADB datasets, which limits the estimates of change available in Table 4-3 (and later in Appendix Table C-6) when both data sources are included in a calculation. Thus, totals calculated this way do not necessarily add to totals calculated as more aggregate stocks. The RPA and FIADB datasets are based on surveys and are explained in detail in Section 4.5.

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). However, this trend is not apparent when comparing regional average values for carbon density (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 apparent regional trends 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

Map 4-3

Regions Used for Carbon Stock Summaries



Regions in the lower map are: Pacific Coast, Rocky Mountain, North and South.

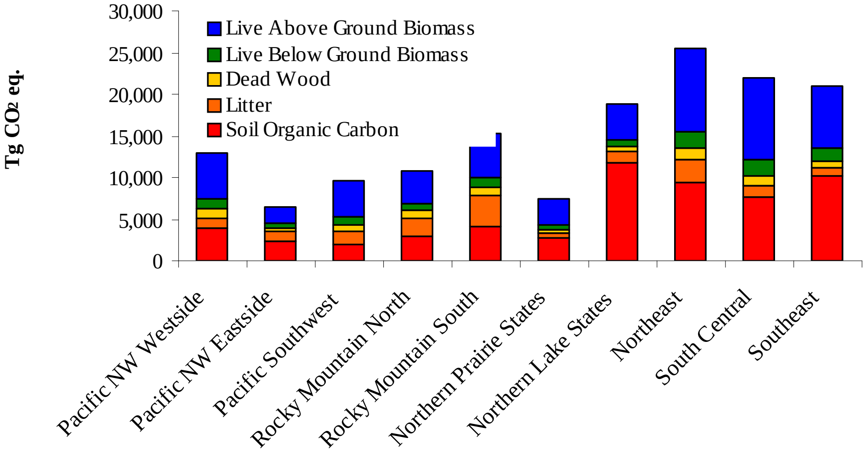
Regions in the upper map are: Pacific Northwest – Westside (PNWW); Pacific Northwest – Eastside (PNWE); Pacific Southwest (PSW); Rocky Mountain – North (RMN); Rocky Mountain – South (RMS); Northern Lake States (NLS); Northern Prairie States (NPS); Northeast (NE); South Central (SC); and Southeast (SE) harvested wood product pools.

Forestland in the conterminous United States is distributed over all 48 of the States. Carbon density of live trees, both above- 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. 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 2005. Carbon stock change in harvested wood is allocated according to total roundwood removals per State from Table 1.10 of Johnson (2001). Calculated values for net annual change reflect estimated carbon densities and forest areas reported in the two most recent surveys per State.

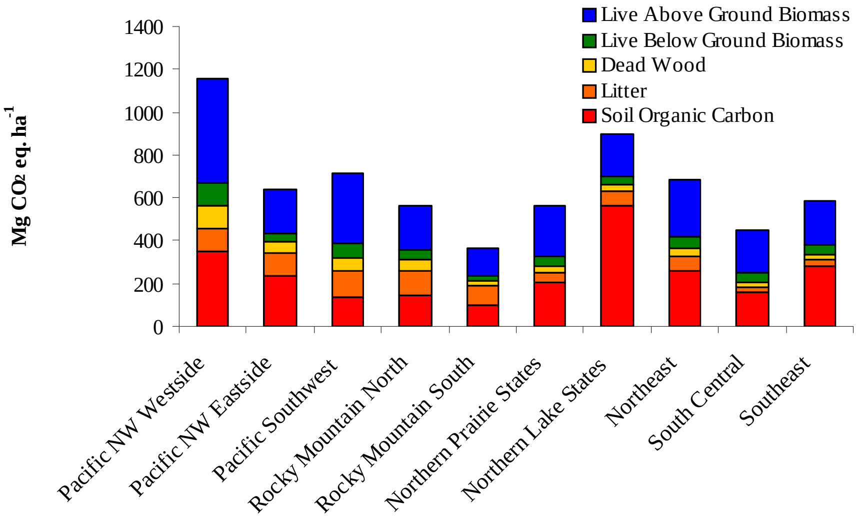
Estimates of net annual change calculated as the difference between two successive inventories are sensitive to changes in forestland over the interval as well as changes in average carbon density. Even small differences in carbon density can contribute to large differences if the change is applied to large areas. Whether change of area or density is the controlling factor is dependent on the situation. Most estimates of net ecosystem carbon change provided in Tables 4-2 and 4-3, Figure 4-2 and Appendix Table C-1 correspond to similar changes in forest area. That is, net gains in forest carbon are most often accompanied by increases in forestland and visa-versa. There are exceptions, and most of these involve net gains in

Figure 4-1

a) **Forest Ecosystem Carbon Stocks**



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



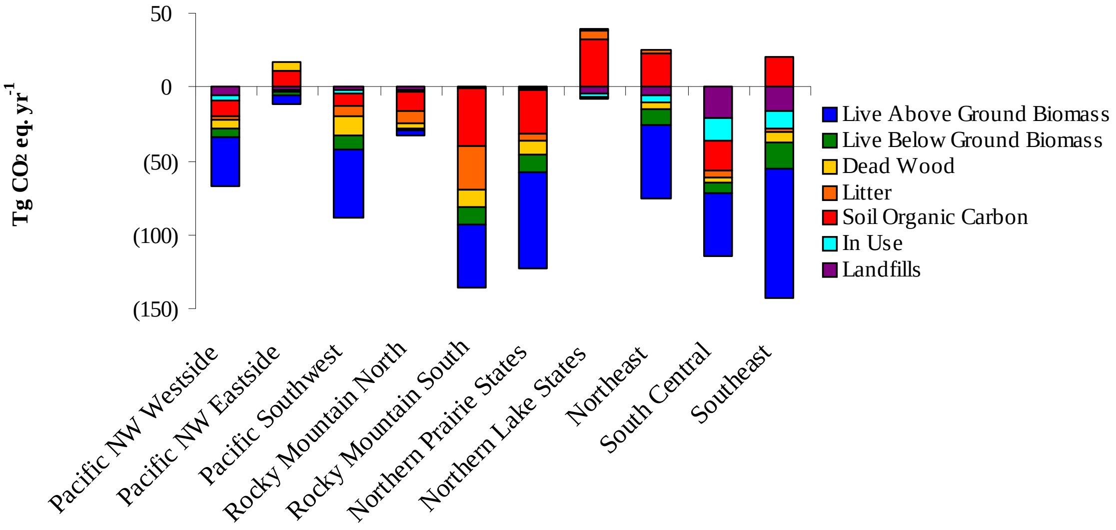
Note: Soil carbon does not include effects of past land use history.

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

forest carbon (negative flux) despite decreases in area. This is the case in Table 4-3 for Eastern White/Red/Jack Pine and Western Douglas-fir, Hemlock/Sitka Spruce, Ponderosa Pine, and Redwood forest type groups. Specifically, each of these type groups decreased in area through the two most recent inventories for their respective locations (data not shown), yet total carbon in biomass increased. Similarly, Appendix Table C-1 shows both of these patterns – carbon stock trend counter to forest area trend – in 12 of the lower 48 States. The four instances of net carbon loss accompanying area gains involve relatively low rates of change (0.2% or less).

Figure 4-2

Net Annual Forest Carbon Stock Change[1]



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

Parentheses indicate a net sequestration. Soil carbon does not include effects of past land use history.

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 while 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 4-2. For more information about forest inventory variables such as forest classifications of ownership, productivity, forest type, and stand size class, see Smith et al. (2004a) and USDA Forest Service (2006).

A large proportion of non-forest trees in the United States are in urban areas – approximately 3% of total tree cover in the conterminous United States (Nowak et al. 2001). 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 2005 is estimated at 88 Tg CO_2 eq. per year (EPA 2007).

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-CO_2 emissions, but the majority of exchange is in terms of CO_2, which is the focus of this chapter. Tree growth results in the net accumulation of CO_2 in forests (removal from the atmosphere), whereas other processes such as respiration, decomposition, or combustion remove CO_2 from the forest. Photosynthesis provides the energy for the conversion of carbon dioxide to organic carbon; this assimilation of CO_2 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 CO_2 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 CO_2 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 (Skog & Nicholson 1998, Smith et al. 2006, Skog in prep).

Forest management affects carbon stocks and stock changes by controlling mechanisms associated with carbon gain and loss (Houghton & Hackler 2000, Johnson & Curtis 2001). For example, increasing tree volume per area of forest generally increases carbon sequestration. 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 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 tool 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).

Harvested wood carbon pools lengthen the time before which carbon returns to the atmosphere. 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 382 Tg CO_2 eq. over the period 1990 through 2005 (EPA 2007, see Table A-199 of Annex 3.12). The newly revised estimates (Skog in prep.) do not specify the portion of emitted carbon that is associated with energy capture (including firewood and wood from products), but previous estimates were about 59% (Skog & Nicholson 1998), which is approximately 225 Tg CO_2 eq. Net annual carbon sequestration via harvested wood, after accounting for these emissions, is specified in Table 4-2.

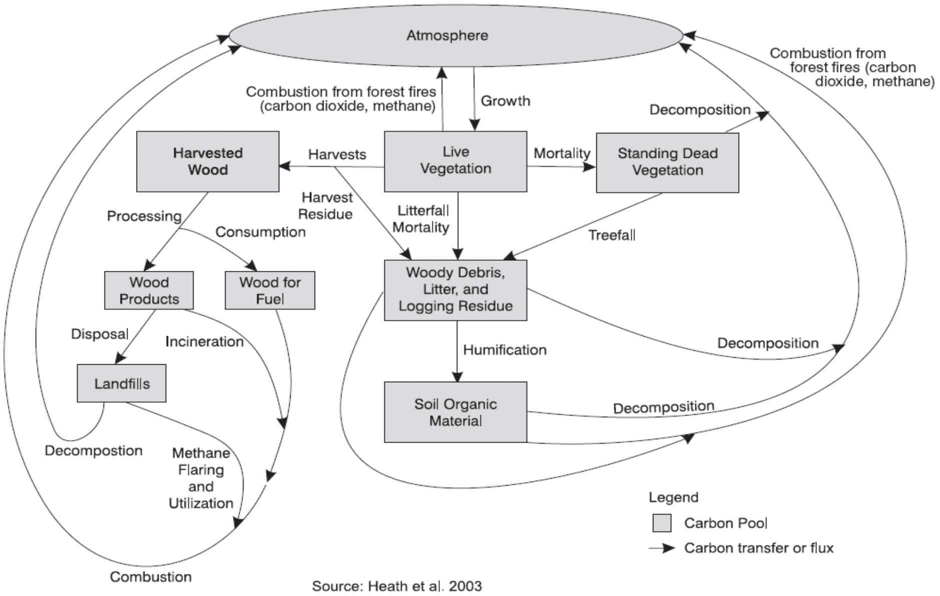
4.5 Methods

The stock change method, which is the basic approach to estimating forest ecosystem carbon as reported here, is to apply factors characteristic of forest carbon pools to inventory data. The USDA Forest Service, Forest Inventory and Analysis (FIA) Program forest inventory data consist of a series of surveys per State, which in the past have been 5 to 10 years apart. The number and frequency of inventories vary from State to State. The new national survey protocol (USDA FS 2006, 2007) calls for a portion of each State to be surveyed each year.

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 (also 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 (USDA 2004); 2) current forest carbon estimates (EPA 2007); 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 (USDA 2004), but the estimates are updated to those in EPA (2007). Classifications, or groupings, of values within tables or figures have changed somewhat due to corresponding changes in forest inventories or carbon pools identified for United Nations Framework Convention on Climate Change (UNFCCC) reporting purposes. Methods are described below with additional details in EPA (2007).

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



Source: Heath et al. 2003

Forest survey

data for the United States are available from the “snapshot” Forest Inventory and Analysis DataBase (FIADB), version 2.1 (USDA FS 2007). 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 these data, both FIADB and RPA, is available on the Internet at the Forest Inventory and Analysis Datacenter (USDA FS 2007). All FIADB surveys used for carbon stock estimates were obtained from the FIADB site on September 8, 2006. See Table 2 of Smith et al. (2007) for a list of the specific surveys, sub-State classifications, and corresponding survey years.

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—or flux—based on the interval (in years) between the stocks. In this way, State-wide annualized estimates of ecosystem stock and flux can be calculated and summed to U.S totals as presented in EPA (2007) and Table 4-2. A similar approach is taken 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 above and belowground portions. For each inventory summary in each State, each carbon pool is estimated using coefficients from the FORCARB2 model (Birdsey & Heath 1995, Birdsey & Heath 2001, Heath et al. 2003, Smith et al. 2004c). Coefficients of the model 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 CO_2 equivalents. The pools are then merged into the set of five reporting pools. FORCARB2’s live tree and understory carbon pools are pooled as biomass in this inventory. Similarly, standing dead trees and down dead wood are pooled as dead wood in this inventory. Definitions of forest floor and SOC in FORCARB2 correspond to litter and forest soils, respectively, as defined in IPCC 2003.

Biomass, or live plant mass, includes trees and understory vegetation. 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 (IPCC 1997). The minimum-sized tree included in these FIA data is one-inch diameter (2.54 cm) at breast height (1.3 meter); this represents the minimum size included in the tree carbon pools. Understory vegetation is defined in FORCARB2 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 IPCC 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).

Dead wood includes the FORCARB2 pools of 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 (Smith et al. 2004c). The standing dead tree carbon pool in FORCARB2 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 standing dead tree carbon are presented in EPA 2007 (Table A-193).

Estimates of forest floor and SOC are not based on carbon density of trees. 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 and 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. 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. Future inventories will include the effects of land use, following the methodology of Woodbury et al. (2007).

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. 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 extrapolating estimates from the last two surveys. Thus, the annualized estimates (based on extrapolation) for 2005 will not exactly match the latest (most recent) data per State. In the results presented in this chapter, all estimates of current (2005) 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 in 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) are based on the methods described by Skog and Nicholson (1998) and substantially revised by Skog (in prep). 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 2005; 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 (in prep). 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 to regions or States is based on estimates in Johnson (2001).

4.6 Major Changes Compared to Previous Inventories

The estimates provided in Table 4-2 reflect two substantial changes between EPA (2006) and EPA (2007) in terms of net stock change since 1990. First, net forest ecosystem carbon sequestration in the early 1990s is revised downward (that is, less negative), and this is accompanied by greater net sequestration in recent years. Thus the overall trend is a shift toward greater carbon accumulation in forests over the interval. This result is not from changes in carbon conversion methods, but rather from availability and resolution of some older inventory data. See Smith et al. (2007) for more discussion of how inventory data were used to develop the current 1990-2005 estimates. For comparison of the respective inventory sets, see Tables A-180 and A-186 of EPA (2006) and EPA (2007), respectively. The significant changes in the use of inventories between the two years were: 1) recognition that “chaparral” was included in older data as a forest type but not in current inventories; 2) the additional sub-State classifications used for identifying sequential sets of carbon stocks; and 3) the addition of the older FIA tree-level data formats. The second substantial change is in the estimates of carbon in harvested wood; see Skog (in prep.) for more information.

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; unknowns or errors in the largely empirical models used to develop the carbon factors; and variability across forests, which are represented by averages. Elements of inventory and carbon conversion unknowns can be addressed separately.

Confidence intervals about volumes and areas are well-defined for forest inventories (Phillips et al. 2000, USDA FS 2006). Additional sources of error in this use of inventory data are related to 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 2- or 3-year period are averaged to a single year. However, if significant portions of a State's forest inventory were sampled on a completely different schedule, 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 (also see EPA 2007 for additional details). The potential for an additional minor error comes from the use of successive surveys and the need for consistent definition, identification and inclusion of all forestlands within a State. If small areas or ownerships are omitted from one of a pair of successive surveys, then a portion of the resulting Statewide flux (net annual 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, as suggested by States with relatively high rates of change in forestland. For example, current calculations for Utah (Table C-1) produce a relative growth rate of over 2% or 173,000 hectares per year, which may be related to definitions of forestlands in successive inventories. Ongoing work will improve resolution of carbon and inventory data.

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 in prep). Additional details on the analysis are provided in Chapter 7 and Annex 3.12 of EPA (2007). The uncertainty analysis was performed using the IPCC-recommended Tier 2 uncertainty estimation methodology, that is, the Monte Carlo simulation technique. The 2005 forest carbon stock changes are estimated to lie between -890 and -513 Tg CO_2 eq. at a 95% confidence level, at a sink of -699 Tg CO_2 eq. (Table 4-1). The 95% confidence intervals for forest sequestration are -785 to -410 Tg CO_2 eq. and -130 to -79 Tg CO_2 eq. for harvested wood products.

4.8 Planned Improvements

The ongoing annualized surveys by the FIA Program will improve precision of forest carbon estimates as new State surveys become available. The annualized surveys will also better reflect the effects of disturbances on forest carbon. In addition, the more intensive sampling of down dead wood, forest floor, and SOC on some of the permanent plots will substantially improve resolution of carbon pools at the plot-level. As more information becomes available about historical land use, the ongoing effects of changes in land use and forest management will be better accounted for in estimates of soil carbon. Urban trees and agroforestry systems represent two broad classes of carbon sequestration by trees that are on lands not currently identified as forest for purposes of the FIA inventories. Estimates of carbon sequestration by urban forests are included in U.S. GHG Inventory (EPA 2007), but future collection of field data (which is underway) as well as reconciling these areas with FIA forest data should improve overall estimates of sequestration. The estimates of carbon stored in harvested wood products are currently being revised using more detailed wood products production and use data, and more detailed parameters on disposition and decay of products.

Agroforestry systems are not currently included in FIA inventory data. Agroforestry practices, such as windbreaks or riparian forest buffers along waterways, are generally not included in forest carbon estimates. Additional research is underway to develop methods for including these systems in nationwide inventories (Perry et al. 2005). This should lead to the inclusion of carbon stock and flux estimates in the forest greenhouse gas inventories. Annual surveys will also eventually include all 50 States. This is particularly important for Alaska which has a large area of forested land.

Chapter 5: Energy Use in Agriculture

5.1 Summary of Greenhouse Gas Emissions from Energy Use in Agriculture

Almost one quadrillion btu of direct energy was used for agriculture in 2005, resulting in about 69 Tg of CO_2 emissions (Table 5-1). The same year, total energy consumption for all sectors in the U.S., including agriculture, was approximately 96 quadrillion btu, resulting in 5943Tg of CO_2 emissions (EPA 2007). Production agriculture's contribution to this total was very small at a little more than 1%. Within agriculture, diesel fuel accounted for about 43% and electricity for about 33% of CO_2 emissions from energy use. Gasoline consumption accounted for about 13% of CO_2 emissions, while LP gas and natural gas accounted for about 7% and 4%, respectively.

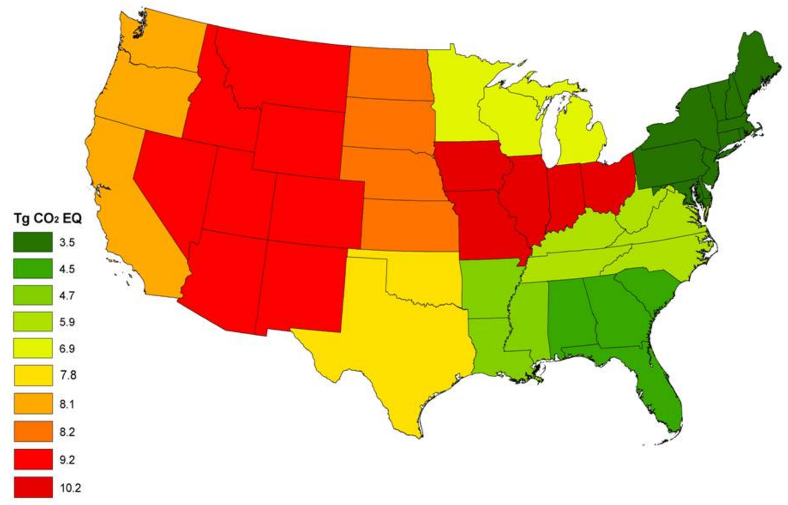
5.2 Spatial and Temporal Trends in Greenhouse Gas Emissions from Energy Use in Agriculture

The highest emissions in 2005 were in the Corn Belt and Mountain States (Map 5-1). Intermediate emissions occurred in the Pacific, Northern Plains, 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 agricultural production use the most energy and therefore have the highest CO_2 emissions. However, emissions also vary by the types of energy used for farm production in each region. For example, even though the Pacific region had the overall highest energy use in 2005, it ranked only fourth in CO_2 emissions, because much of the energy used for agricultural production in the Pacific region comes from hydroelectric power.

Agricultural energy use and resulting CO_2 emissions grew throughout the 1960s and 1970s, peaking in the late 1970s (Figure 5-1). High prices, stemming from the oil crisis of the 1970s and early 1980s, drove farmers to be more energy-efficient, driving 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, USDA ERS 1994, 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

Map 5-1

Farm Energy Use by Region in 2005



throughout the 1980s and 1990s.

Farm energy use leveled off in the late 1980s as energy prices subsided (Figure 5-1). Since 1990 there has been an upward movement in energy use; however, farm energy used today is still well below the peak levels that occurred in the 1970s. Moreover, energy production, energy output per unit of energy input, has increased significantly.

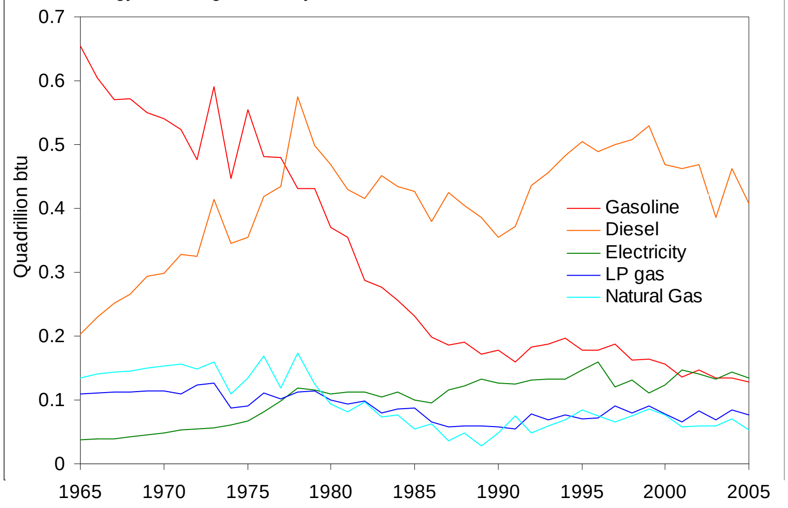


Figure 5-1
Energy Use in Agriculture by Source, 1965-2005

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

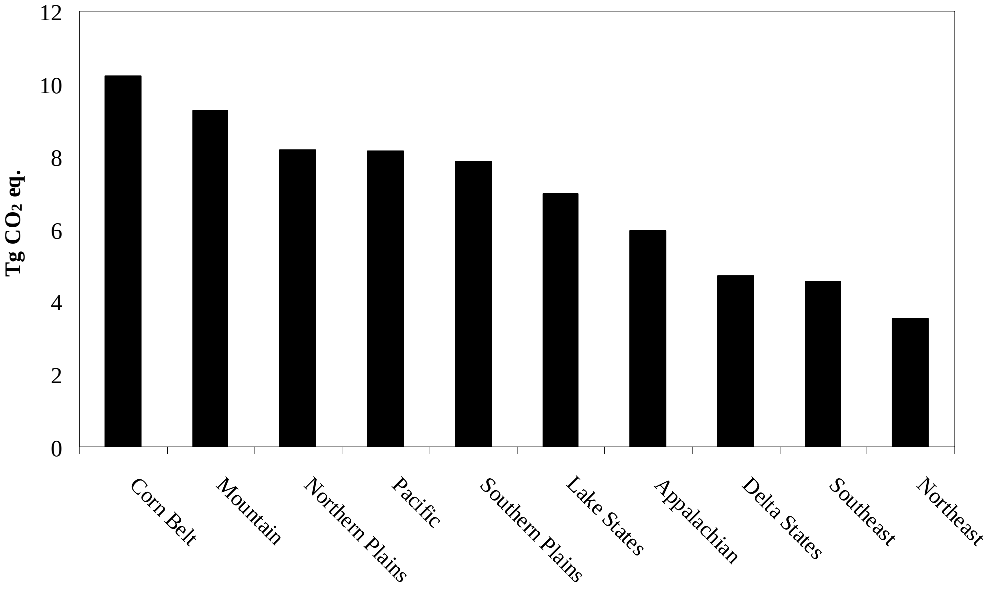
Agricultural operations, including crop and livestock farms, dairies, nurseries and greenhouses, require a variety of energy sources. Energy use in agriculture varies across agricultural operations by crop or livestock type, size of operation, and geographic region (Figure 5-2, Table 5-1). Energy use 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 (Maranowski 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.

Figure 5-2

Carbon Dioxide Emissions from Energy Use in Agriculture by Region[1] in 2005



[1] Regions are defined in Appendix Table D-1.

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 2003 USDA Farm and Ranch Irrigation Survey, about 43 million acres of U.S. farmland were irrigated with pumps powered by liquid fuels, natural gas and electricity, costing a total of $1.55 billion (USDA NASS 2004). Electricity was the principal power source for these pumps, costing $953 million to irrigate 24.1 million acres at an average cost of $39.50 per acre. Diesel fuel was used to power pumps on about 12 million acres and natural gas was used on about 5 million acres (USDA NASS 2004).

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

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				

The area of land irrigated can vary substantially from year to year, depending on environmental conditions. For example, in 2003, 52.6 million acres of farmland in the U.S. were irrigated (including gravity flow irrigation), about 6 million acres more than were irrigated in 1994 (USDA NASS 1999d, 2004). Corn for grain or seed, alfalfa hay, cotton, soybeans, and orchard land (e.g., fruit trees, vineyards, and nut trees) required the most water in 2003, accounting for 56% of all irrigated land. The leading States for irrigated land in 2003 are California with 16%, Nebraska with 14%, and Texas with 9%, out of the total U.S. irrigated farm land area.

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 Inventory of U.S. Greenhouse Gas Emissions and Sinks (EPA 2007).

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 5-3). Some fuels have higher carbon content than others, resulting in higher CO_2 emissions per btu 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 than 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

Carbon dioxide emission estimates for energy use are constructed from fuel consumption data using standardized methods published in the U.G. GHG Inventory (EPA 2007). Emission estimates from fuel use in agriculture are not explicitly published in the U.S. GHG Inventory; 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. GHG Inventory.

Estimates of CO_2 from agricultural operations are based on energy 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 2006). NASS also collects data on price per gallon paid by farmers for gasoline, diesel, and LP gas (USDA NASS 2005a). 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 is aggregated by State and the State data is 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), which reports average prices by State (EIA 2005a, 2005b). 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. GHG Inventory (EPA 2007), consumption of diesel fuel, gasoline, LP gas and natural gas was converted to CO_2 emissions using the coefficients for carbon content of fuels and fraction of carbon oxidized during combustion, both of which are published in Annex 2 and provided in Table 5-1 of this report. 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 2007). However, only a portion of the carbon is actually oxidized during combustion. These coefficients are provided in Table 5-1 of this report. It is assumed that of the carbon that is oxidized, 100% is emitted to the atmosphere as CO_2.

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 over time. 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. EIA typically reports CO_2 emissions from electricity generation by State and U.S. Census Regions (EIA 2001). 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

Table 5-2 Energy Use and Carbon Dioxide Emissions by Fuel Source on U.S. Farms, 2005

	Energy Consumed		**Carbon Content**	**Fraction Oxidized**	**CO_2 Emissions**
	Trillion Btu's	*Qbtu*	*Tg C/Qbtu*		*Tg CO_2 eq.*
Fuels					
Diesel	408.5	0.4085	19.95	0.99	29.58
Gasoline	128.5	0.1285	19.33	0.99	9.01
LP gas	76	0.076	17.2	0.99	4.74
Natural gas	53	0.053	14.47	0.995	2.80
Electricity	135	0.135	*	*	23.28
Total	**801**	**0.801**			**69.41**

* Varies depending on fuel used to generate electricity and heat rate of the power generating facility.

Note: The BTUs for electricity consumed are based on 3,413 BTU per kWh, which is just the direct energy used on the farm. The emission coeficients from EIA include the energy source, e.g. the emissions from electricity produced from coal are calculated upstream at the coal-fired plant thus the estimated emissions for electricity are much greater than just the emissions from using electricity on the farm.

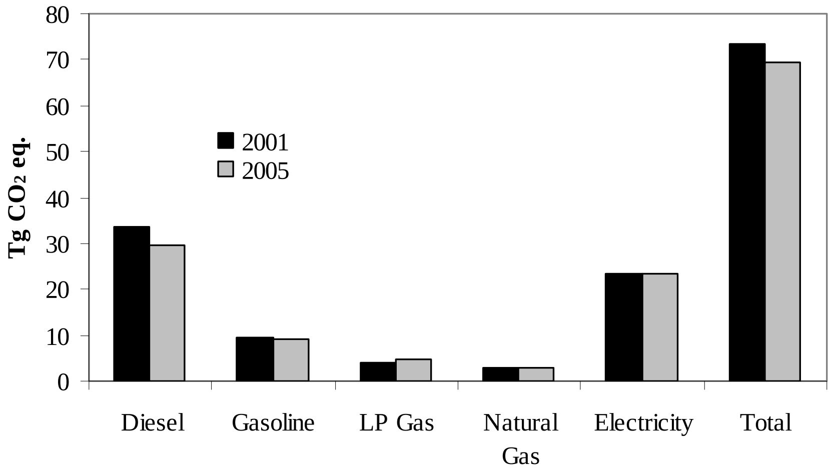
use is estimated from farm expenditure data collected by NASS. Price estimates for electricity published by EIA are divided into electricity expenditures to derive the kilowatt hours consumed by farmers. The kilowatt hours of electricity are converted to btu, based on a conversion rate of 3,413 btu per kilowatt hour.

5.5. Major Changes Compared to Previous Inventories

The first edition of the USDA GHG report (USDA 2004) included estimates of emissions from energy use for the year 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. Figure 5-3 shows that the results from the two study years are very similar. The total 2005 CO_2 emissions from energy production in agriculture are only about 4 Tg of CO_2 lower than the emissions estimated for 2001. The lower emissions in 2005 result primarily from farmers using less diesel fuel in 2005.

Figure 5-3

Carbon Dioxide Emissions from Energy Use in Agriculture by Fuel Source, 2001 & 2005



Note that the 2001 CO_2 emission estimates have been revised with updated data, so they are different than the estimates reported in the first edition of this report. As is often the case with survey data, the initial 2001 data used to derive the energy use estimates have been updated. With the exception of electricity, the revised 2001 CO_2 emissions estimates are very similar to the estimates originally reported. However, due to a calculation error, the 2001 figure reported for electricity in the original study was overestimated. Consequently, the revised CO_2 emission estimate for electricity is significantly lower than that reported previously.

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Appendix A: Livestock Emissions

A-1   Population of Animals by State in 2005
A-2   U.S. Livestock Population, 1990-2005
A-3   State-Level Methane Emissions from Enteric Fermentation in 1990-2005
A-4   State-Level Methane Emissions from Enteric Fermentation by Livestock Category in 2005
A-5   Cattle Population Categories Used for Estimating Methane Emissions
A-6   Dairy Lactation by Region
A-7   Typical Livestock Weights
A-8   U.S. Feedlot Placement in 2005
A-9   Regional Estimates of Digestible Energy and Methane Conversion Rates for Enteric Fermentation
A-10 Definition of Regions in the Enteric Fermentation Model
A-11 Methane Emissions from Cattle Enteric Fermentation, 1990-2005
A-12 IPCC Emission Factors for Livestock
A-13 Summary of Greenhouse Gas Emissions from Managed Waste by State
A-14 Methane Emissions from Manure Management by State and Animal in 2005
A-15 Nitrous Oxide Emissions from Manure Management by State and Animal in 2005
A-16 Waste Characteristics Data
A-17 State Volatile Solids Production Rates in 2005
A-18 State-Based Methane Conversion Factors by Waste Management System in 2005
A-19 Additional Nitrous Oxide and Methane Emission Factors
A-20 State-Weighted Methane Conversion Factors for Livestock Waste Emissions in 2005
A-21 Nitrogen in Livestock Waste on Grazed Lands

Appendix Table A-1 Population of Animals by State in 2005

	Beef Cattle	Dairy Cattle	Swine	Sheep	Goat	Horse	Poultry
State				*Head*			
Alabama	1,488,231	24,645	180,000	9,000	50,574	95,250	205,538,724
Alaska	12,839	1,675	1,700	9,000	277	2,795	948,000
Arizona	728,823	190,158	136,000	114,000	35,374	68,159	948,000
Arkansas	2,028,200	37,494	325,000	9,000	32,580	105,115	258,289,270
California	3,178,735	2,392,859	140,000	680,000	103,122	191,901	48,152,594
Colorado	2,188,237	140,706	740,000	360,000	18,561	155,746	5,761,302
Connecticut	27,134	30,963	4,200	7,167	2,586	13,815	3,618,498
Deleware	15,557	10,659	15,000	9,000	1,521	5,022	45,481,936
Florida	1,720,948	177,992	20,000	9,000	39,964	145,304	27,359,730
Georgia	1,281,592	108,691	275,000	9,000	69,498	107,632	265,515,640
Hawaii	162,904	7,900	22,000	9,000	5,364	6,664	597,996
Idaho	1,470,310	582,838	21,000	260,000	11,520	121,033	1,247,005
Illinois	1,220,447	157,347	3,962,500	63,000	17,192	86,750	24,664,590
Indiana	693,035	195,211	3,175,000	50,000	27,801	143,098	53,551,586
Iowa	3,232,628	286,151	16,050,000	250,000	18,898	112,163	79,080,596
Kansas	6,467,615	185,885	1,737,500	100,000	24,763	98,063	1,718,300
Kentucky	2,365,975	149,975	350,000	26,000	68,412	217,372	59,830,731
Louisiana	865,129	50,502	16,000	9,000	14,633	69,500	2,573,997
Maine	45,549	51,554	5,000	7,167	3,162	18,525	4,250,995
Maryland	144,092	105,478	26,000	25,000	9,601	37,740	56,611,457
Massachusetts	24,979	26,541	12,000	7,167	6,022	22,539	327,995
Michigan	668,630	423,384	930,000	83,000	21,094	152,631	29,682,591
Minnesota	1,818,803	730,645	6,550,000	140,000	19,768	134,919	36,833,186
Mississippi	1,074,181	40,390	315,000	9,000	26,738	97,170	161,657,090
Missouri	4,522,610	181,939	2,912,500	60,000	48,654	205,588	34,730,594
Montana	2,351,195	25,593	165,000	300,000	8,613	137,283	480,000
Nebraska	5,993,940	80,991	2,850,000	102,000	11,718	85,701	15,524,115
Nevada	504,961	34,493	5,500	75,000	6,506	23,448	948,000
New Hampshire	18,145	23,591	3,600	7,167	3,774	11,527	252,198
New Jersey	30,957	17,694	11,000	9,000	8,312	39,116	2,095,098
New Mexico	1,186,612	396,272	2,500	160,000	19,128	67,897	948,000
New York	567,912	942,712	84,000	70,000	33,130	109,468	6,144,725
North Carolina	891,708	83,624	10,050,000	20,000	67,276	93,351	160,365,455
North Dakota	1,697,117	46,288	169,000	100,000	2,523	63,140	1,248,000
Ohio	941,581	367,144	1,487,500	140,000	45,061	195,416	45,333,636
Oklahoma	5,274,643	99,002	2,412,500	75,000	82,792	218,357	50,537,573
Oregon	1,304,208	183,411	27,000	215,000	30,628	134,415	23,286,889
Pennsylvania	939,772	829,702	1,045,000	90,000	39,932	164,922	55,804,732
Rhode Island	3,810	1,964	2,000	7,167	468	2,843	948,000
South Carolina	450,832	23,644	300,000	9,000	41,192	59,226	47,824,820
South Dakota	3,571,022	112,218	1,257,500	370,000	7,021	101,175	5,073,004
Tennessee	2,331,568	110,217	215,000	22,000	114,664	216,191	38,018,180
Texas	14,227,217	421,439	955,000	1,100,000	1,194,289	541,508	138,184,844
Utah	765,189	126,962	690,000	265,000	9,092	89,250	4,647,302
Vermont	93,688	209,585	2,000	7,167	4,133	16,351	240,603
Virginia	1,549,754	141,076	375,000	55,000	41,275	118,301	58,634,184
Washington	818,636	330,177	26,000	46,000	23,217	110,458	24,779,588
West Virginia	385,280	17,799	10,000	34,000	17,484	46,325	18,932,092
Wisconsin	1,697,521	1,880,727	440,000	83,000	35,179	148,336	13,395,760
Wyoming	1,404,635	6,845	114,000	430,000	5,380	91,501	17,000
Total	**86,449,086**	**12,804,752**	**60,620,500**	**6,105,000**	**2,530,466**	**5,300,000**	**2,122,636,203**

Source: EPA 2007

Appendix Table A-2 U.S. Livestock Population, 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Animal Type								*1,000,000 head*								
Dairy Cattle	**2**	**14**	**14**	**14**	**14**	**14**	**13**	**13**	**13**	**13**	**13**	**13**	**13**	**13**	**13**	**13**
Dairy Cows	10	10	10	10	10	9	9	9	9	9	9	9	9	9	9	9
Dairy Heifers	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Swine	**54**	**56**	**59**	**58**	**60**	**59**	**56**	**59**	**62**	**60**	**59**	**59**	**60**	**60**	**61**	**61**
Market <60 lbs.	18	19	20	19	20	20	19	20	21	20	20	20	20	20	20	20
Market 60-119 lbs.	12	12	13	13	13	13	12	13	14	13	13	13	13	13	13	14
Market 120-179 lbs.	10	10	10	10	11	11	10	10	11	11	11	11	11	11	11	11
Market >180 lbs.	8	8	8	8	9	9	8	9	10	10	9	9	10	10	10	10
Breeding Swine	7	7	7	7	7	7	7	7	7	6	6	6	6	6	6	6
Beef cattle	**86**	**87**	**89**	**90**	**93**	**94**	**94**	**92**	**91**	**90**	**89**	**89**	**88**	**87**	**86**	**87**
Feedlot Steers	7	8	8	8	8	8	7	8	8	8	8	9	8	8	8	8
Feedlot Heifers	4	4	4	4	4	4	4	4	4	5	5	5	5	5	5	5
Bulls NOF[1]	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Calves NOF	24	24	24	24	25	25	25	24	24	24	24	23	23	22	22	22
Heifers NOF	9	9	10	10	10	11	11	10	10	10	9	9	9	9	9	9
Steers NOF	7	7	8	8	8	9	9	8	8	8	7	7	7	7	7	7
Cows NOF	33	33	33	34	35	36	36	35	34	34	34	34	33	33	33	33
Sheep	**11**	**11**	**11**	**10**	**10**	**9**	**8**	**8**	**8**	**7**	**7**	**7**	**7**	**6**	**6**	**6**
Goats	**3**	**3**	**3**	**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**	**2**	**3**	**3**	**3**	**3**
Poultry	**1,537**	**1,595**	**1,650**	**1,707**	**1,769**	**1,827**	**1,882**	**1,927**	**1,965**	**2,009**	**2,033**	**2,060**	**2,098**	**2,085**	**2,131**	**2,151**
Hens >1 yr.	273	280	285	291	299	299	304	312	322	330	334	340	340	341	344	348
Pullets	73	77	80	82	80	81	82	90	96	98	95	96	95	100	101	97
Chickens	7	7	7	7	7	8	7	8	8	10	8	8	8	8	8	8
Broilers	1,066	1,116	1,164	1,217	1,276	1,332	1,381	1,412	1,443	1,481	1,506	1,525	1,562	1,544	1,589	1,613
Turkeys	118	116	114	111	107	107	108	105	97	90	90	91	92	91	88	85
Horses	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**	**5**

Source: EPA 2007

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 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2004	2005
State							*Tg CO_2 eq.*							
Alabama	1.79	1.74	1.77	1.78	1.84	1.94	1.91	1.74	1.70	1.64	1.62	1.51	1.58	1.50
Alaska	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.02	0.02	0.01	0.02	0.02	0.02
Arizona	1.10	1.10	1.15	1.14	1.14	1.12	1.11	1.07	1.11	1.11	1.08	1.08	1.09	1.14
Arkansas	2.05	1.95	1.97	1.95	2.12	2.24	2.17	2.15	2.06	2.05	2.07	2.07	2.17	2.07
California	6.61	6.42	6.51	6.17	6.44	6.50	6.06	6.21	6.20	6.50	6.59	6.72	7.53	7.80
Colorado	3.15	2.99	3.26	3.18	3.20	3.22	3.25	3.30	3.32	3.26	3.18	3.12	2.41	2.49
Connecticut	0.12	0.12	0.12	0.12	0.12	0.12	0.11	0.11	0.11	0.11	0.11	0.10	0.09	0.09
Deleware	0.04	0.04	0.05	0.05	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Florida	2.52	2.53	2.54	2.52	2.59	2.64	2.50	2.45	2.34	2.28	2.29	2.26	2.30	2.23
Georgia	1.64	1.66	1.67	1.65	1.69	1.72	1.65	1.60	1.50	1.47	1.48	1.44	1.55	1.49
Hawaii	0.25	0.27	0.24	0.23	0.22	0.23	0.22	0.23	0.24	0.23	0.22	0.20	0.19	0.18
Idaho	2.26	2.30	2.38	2.33	2.37	2.51	2.52	2.56	2.63	2.67	2.77	2.83	2.64	2.69
Illinois	2.12	2.09	2.14	2.14	2.08	2.00	1.87	1.82	1.80	1.82	1.80	1.74	1.59	1.64
Indiana	1.50	1.44	1.42	1.37	1.42	1.42	1.34	1.34	1.27	1.28	1.23	1.16	1.18	1.21
Iowa	5.05	4.95	5.01	4.79	4.67	4.75	4.43	4.32	4.21	4.34	4.27	4.15	4.05	4.13
Kansas	5.70	5.56	5.55	5.62	5.70	6.01	5.95	5.94	5.82	5.97	5.94	5.98	5.81	5.75
Kentucky	2.89	2.94	3.01	2.97	2.99	3.03	3.00	2.90	2.80	2.69	2.52	2.57	2.82	2.70
Louisiana	1.34	1.28	1.21	1.24	1.20	1.22	1.20	1.19	1.17	1.14	1.14	1.10	1.10	1.07
Maine	0.17	0.17	0.17	0.17	0.17	0.16	0.16	0.16	0.16	0.16	0.15	0.15	0.14	0.14
Maryland	0.47	0.47	0.45	0.46	0.45	0.45	0.42	0.39	0.38	0.38	0.37	0.35	0.36	0.36
Massachusetts	0.13	0.12	0.13	0.12	0.12	0.11	0.11	0.10	0.10	0.10	0.10	0.08	0.08	0.08
Michigan	1.52	1.47	1.49	1.50	1.52	1.52	1.46	1.41	1.37	1.37	1.34	1.31	1.43	1.46
Minnesota	3.37	3.42	3.44	3.45	3.25	3.38	3.34	3.30	3.18	3.15	3.20	3.16	3.01	3.02
Mississippi	1.51	1.50	1.52	1.57	1.56	1.51	1.53	1.43	1.36	1.31	1.24	1.25	1.24	1.25
Missouri	5.08	5.10	5.14	5.24	5.46	5.42	5.39	5.32	5.11	5.10	4.98	4.94	5.06	4.99
Montana	3.11	3.25	3.49	3.44	3.43	3.63	3.60	3.56	3.49	3.49	3.53	3.48	3.07	2.93
Nebraska	5.74	5.84	5.91	5.82	6.10	6.21	6.43	6.49	6.62	6.69	6.58	6.39	5.90	5.99
Nevada	0.69	0.69	0.72	0.70	0.69	0.71	0.68	0.71	0.70	0.70	0.72	0.73	0.62	0.60
New Hampshire	0.07	0.08	0.08	0.08	0.08	0.08	0.07	0.07	0.07	0.08	0.08	0.07	0.07	0.07
New Jersey	0.13	0.13	0.13	0.12	0.12	0.11	0.11	0.11	0.10	0.10	0.09	0.09	0.08	0.08
New Mexico	1.91	1.88	1.92	1.99	2.03	2.12	2.22	2.24	2.30	2.33	2.35	2.35	2.15	2.10
New York	2.49	2.50	2.54	2.43	2.40	2.37	2.27	2.29	2.32	2.34	2.35	2.26	2.35	2.35
North Carolina	1.15	1.22	1.29	1.31	1.39	1.49	1.53	1.52	1.39	1.40	1.38	1.39	1.38	1.34
North Dakota	2.19	2.25	2.28	2.34	2.46	2.53	2.42	2.41	2.29	2.39	2.40	2.55	2.12	2.07
Ohio	1.96	1.95	1.92	1.81	1.78	1.79	1.79	1.70	1.67	1.60	1.59	1.59	1.67	1.75
Oklahoma	5.49	5.53	5.59	5.50	5.40	5.83	5.70	5.57	5.63	5.49	5.48	5.40	5.37	5.48
Oregon	2.02	2.02	2.01	1.99	2.11	2.21	2.24	2.22	2.17	2.15	2.11	2.01	1.78	1.80
Pennsylvania	2.67	2.66	2.74	2.60	2.53	2.53	2.45	2.43	2.42	2.40	2.39	2.37	2.40	2.43
Rhode Island	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
South Carolina	0.67	0.59	0.62	0.63	0.61	0.61	0.61	0.59	0.56	0.56	0.55	0.53	0.54	0.53
South Dakota	4.20	4.18	4.35	4.62	4.49	4.80	4.59	4.47	4.32	4.52	4.62	4.91	4.21	4.20
Tennessee	2.61	2.59	2.59	2.58	2.72	2.80	2.76	2.56	2.48	2.38	2.35	2.39	2.61	2.50
Texas	14.78	14.53	15.27	15.53	16.22	16.47	16.11	15.16	15.12	15.03	14.59	14.33	14.80	14.39
Utah	1.23	1.24	1.23	1.30	1.30	1.34	1.36	1.39	1.39	1.35	1.40	1.42	1.15	1.13
Vermont	0.52	0.51	0.51	0.51	0.49	0.50	0.48	0.48	0.50	0.51	0.50	0.49	0.49	0.49
Virginia	1.97	1.98	2.00	1.94	1.93	1.96	1.95	1.95	1.86	1.84	1.76	1.81	1.86	1.90
Washington	1.96	1.99	1.98	2.00	2.02	1.94	1.85	1.79	1.76	1.73	1.77	1.70	1.51	1.43
West Virginia	0.57	0.59	0.60	0.59	0.58	0.59	0.57	0.53	0.52	0.51	0.49	0.47	0.47	0.48
Wisconsin	5.64	5.59	5.51	5.36	5.13	5.11	4.98	4.88	4.85	4.79	4.83	4.77	4.83	4.89
Wyoming	1.66	1.66	1.78	1.86	2.00	1.95	1.94	2.09	2.13	2.01	2.03	1.98	1.71	1.66
Total	**117.85**	**117.10**	**119.39**	**118.82**	**120.40**	**122.96**	**120.47**	**118.33**	**116.70**	**116.58**	**115.68**	**114.82**	**112.61**	**112.13**

Source: EPA 2007

Appendix Table A-4 State-Level Methane Emissions from Enteric Fermentation by Livestock Category in 2005

	Beef cattle	Dairy cattle	Swine	Sheep	Goats	Horses	***Total***
State			*Tg C0$_2$ eq.*				
Alabama	1.405	0.049	0.005	0.002	0.005	0.036	**1.50**
Alaska	0.013	0.004	0.000	0.002	0.000	0.001	**0.02**
Arizona	0.581	0.507	0.004	0.017	0.004	0.026	**1.14**
Arkansas	1.932	0.080	0.009	0.002	0.003	0.040	**2.07**
California	2.251	5.350	0.005	0.113	0.011	0.073	**7.80**
Colorado	2.029	0.311	0.026	0.061	0.002	0.059	**2.49**
Connecticut	0.020	0.061	0.000	0.001	0.000	0.005	**0.09**
Deleware	0.012	0.025	0.001	0.002	0.000	0.002	**0.04**
Florida	1.739	0.424	0.001	0.002	0.004	0.055	**2.23**
Georgia	1.174	0.258	0.009	0.002	0.007	0.041	**1.49**
Hawaii	0.160	0.018	0.001	0.002	0.001	0.003	**0.18**
Idaho	1.262	1.338	0.001	0.045	0.001	0.046	**2.69**
Illinois	1.145	0.323	0.127	0.012	0.002	0.033	**1.64**
Indiana	0.569	0.477	0.097	0.008	0.003	0.054	**1.21**
Iowa	2.952	0.575	0.513	0.041	0.002	0.042	**4.13**
Kansas	5.297	0.338	0.054	0.018	0.003	0.037	**5.75**
Kentucky	2.255	0.338	0.012	0.005	0.007	0.082	**2.70**
Louisiana	0.930	0.111	0.000	0.002	0.002	0.026	**1.07**
Maine	0.032	0.101	0.000	0.001	0.000	0.007	**0.14**
Maryland	0.111	0.224	0.001	0.004	0.001	0.014	**0.36**
Massachusetts	0.018	0.052	0.000	0.001	0.001	0.009	**0.08**
Michigan	0.408	0.944	0.029	0.014	0.002	0.058	**1.46**
Minnesota	1.333	1.414	0.207	0.024	0.002	0.051	**3.02**
Mississippi	1.118	0.080	0.012	0.002	0.003	0.037	**1.25**
Missouri	4.441	0.366	0.089	0.011	0.005	0.078	**4.99**
Montana	2.768	0.366	0.006	0.051	0.001	0.052	**2.93**
Nebraska	5.666	0.188	0.088	0.016	0.001	0.032	**5.99**
Nevada	0.498	0.077	0.000	0.012	0.001	0.009	**0.60**
New Hampshire	0.012	0.049	0.000	0.001	0.000	0.004	**0.07**
New Jersey	0.024	0.037	0.000	0.002	0.001	0.015	**0.08**
New Mexico	1.070	0.978	0.000	0.024	0.002	0.026	**2.10**
New York	0.293	1.999	0.003	0.013	0.003	0.041	**2.35**
North Carolina	0.815	0.169	0.311	0.003	0.007	0.035	**1.34**
North Dakota	1.922	0.101	0.005	0.018	0.000	0.024	**2.07**
Ohio	0.781	0.818	0.049	0.024	0.005	0.074	**1.75**
Oklahoma	5.075	0.231	0.075	0.012	0.009	0.083	**5.48**
Oregon	1.336	0.369	0.001	0.038	0.003	0.051	**1.80**
Pennsylvania	0.570	1.740	0.034	0.017	0.004	0.062	**2.43**
Rhode Island	0.004	0.003	0.000	0.001	0.000	0.001	**0.01**
South Carolina	0.438	0.055	0.010	0.002	0.004	0.022	**0.53**
South Dakota	3.812	0.246	0.044	0.063	0.001	0.038	**4.20**
Tennessee	2.174	0.221	0.006	0.004	0.012	0.082	**2.50**
Texas	12.873	0.978	0.030	0.180	0.125	0.205	**14.39**
Utah	0.761	0.271	0.022	0.045	0.001	0.034	**1.13**
Vermont	0.043	0.440	0.000	0.001	0.000	0.006	**0.49**
Virginia	1.507	0.323	0.015	0.010	0.004	0.045	**1.90**
Washington	0.656	0.723	0.001	0.008	0.002	0.042	**1.43**
West Virginia	0.410	0.040	0.000	0.005	0.002	0.018	**0.48**
Wisconsin	1.006	3.798	0.013	0.014	0.004	0.056	**4.89**
Wyoming	1.529	0.012	0.004	0.076	0.001	0.035	**1.66**
Total	**79.23**	**27.69**	**1.92**	**1.03**	**0.27**	**2.00**	**112.13**

Source: EPA 2007

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 2007

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

	California	West	Northern Great Plains	Southcentral	Northeast	Midwest	Southeast
Year				*(lbs * year)/cow*			
1990	18,443	17,293	13,431	13,399	14,557	14,214	12,852
1991	18,522	17,615	13,525	13,216	14,985	14,446	13,053
1992	18,709	18,083	13,998	13,656	15,688	14,999	13,451
1993	18,839	18,253	14,090	14,027	15,602	15,086	13,739
1994	20,190	18,802	14,686	14,395	15,732	15,276	14,111
1995	19,559	18,708	14,807	14,294	16,254	15,680	14,318
1996	19,148	19,076	15,040	14,402	16,271	15,651	14,232
1997	19,815	19,537	15,396	14,330	16,519	16,116	14,517
1998	19,437	19,814	15,919	14,722	16,864	16,676	14,404
1999	20,767	20,477	16,325	14,990	17,246	16,966	14,840
2000	21,116	20,781	17,205	15,363	17,482	17,426	15,176
2001	20,890	20,775	17,242	14,952	17,603	17,217	15,304
2002	21,263	21,073	18,079	15,746	18,001	17,576	15,451
2003	20,979	21,132	18,550	16,507	17,727	18,048	15,113
2004	21,125	21,140	18,746	17,567	17,720	18,176	15,696
2005	21,389	21,742	19,627	18,589	18,446	18,839	16,045

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

[1] Beef lactation data developed using methodology described in EPA 2007.

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 2007.

Appendix Table A-8 U.S. Feedlot Placement in 2005

	Jan	Feb	Mar	Apr	May	Jun	July	Aug	Sep	Oct	Nov	Dec	***Total***
Weight Placed	*Number of animals placed, 1,000 head*												
< 600 lbs.	367	319	347	315	495	460	445	506	628	912	590	465	**5,849**
600 - 700 lbs.	466	351	347	304	493	359	324	416	475	764	557	558	**5,414**
700 - 800 lbs.	579	548	646	566	772	453	499	565	552	529	326	489	**6,524**
> 800 lbs.	342	394	470	415	610	375	451	615	720	496	270	322	**5,480**
Total	**1,754**	**1,612**	**1,810**	**1,600**	**2,370**	**1,647**	**1,719**	**2,102**	**2,375**	**2,701**	**1,743**	**1,834**	**23,267**

Source: USDA (2002f, 2001f, 2000f, 1999a, 1995a), EPA 2007.

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

	Data	California	West	Northern Great Plains	Southcentral	Northeast	Midwest	Southeast
Livestock Category								
Beef Replacement Heifer	DE[1]	65	59	66	64	65	65	64
	Y_m[2]	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
	Pop.[3]	3%	10%	31%	23%	2%	14%	17%
Dairy Replacement Heifer	DE	66	66	66	64	68	66	66
	Y_m	5.9%	5.9%	5.6%	6.4%	6.3%	5.6%	6.9%
	Pop.	18%	12%	5%	4%	18%	36%	7%
Steer Stockers	DE	65	59	66	64	65	65	64
	Y_m	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
	Pop.	4%	8%	42%	22%	2%	18%	5%
Heifer Stockers	DE	65	59	66	64	65	65	64
	Y_m	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
	Pop.	2%	7%	50%	22%	1%	15%	4%
Steer Feedlot	DE	85	85	85	85	85	85	85
	Y_m	3%	3%	3%	3%	3%	3%	3%
	Pop.	3%	8%	48%	24%	1%	16%	1%
Heifer Feedlot	DE	85	85	85	85	85	85	85
	Y_m	3%	3%	3%	3%	3%	3%	3%
	Pop.	3%	8%	48%	24%	1%	16%	1%
Beef Cows	DE	63	57	64	62	63	63	62
	Y_m	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%	6.5%
	Pop.	2%	8%	28%	26%	2%	14%	19%
Dairy Cows	DE	69	66	69	68	69	69	68
	Y_m	4.8%	5.8%	5.8%	5.7%	5.8%	5.8%	5.6%
	Pop.	17%	13%	5%	6%	18%	33%	8%
Steer Step-Up	DE	73	73	73	73	73	73	73
	Y_m	4.8%	4.8%	4.8%	4.8%	4.8%	4.8%	4.8%
Heifer Step-Up	DE	73	73	73	73	73	73	73
	Y_m	4.8%	4.8%	4.8%	4.8%	4.8%	4.8%	4.8%

Source: EPA 2007

[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.

[3] (Pop.) Percent of each subcategory population present in each region.

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
Minnesotta	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 2007

Appendix Table A-11 Methane Emissions from Cattle Enteric Fermentation, 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Animal Type	$Gg\ CH_4$															
Dairy	**1,375**	**1,378**	**1,375**	**1,316**	**1,314**	**1,320**	**1,254**	**1,255**	**1,251**	**1,265**	**1,283**	**1,280**	**1,288**	**1,299**	**1,285**	**1,319**
Cows	1,142	1,148	1,143	1,082	1,082	1,088	1,024	1,028	1,026	1,037	1,058	1,053	1,060	1,070	1,058	1,086
Replacements 7-11 mo.	49	49	49	49	49	49	48	48	48	48	48	48	49	48	48	50
Replacements 12-23 mo.	184	181	183	185	183	183	181	179	177	180	177	179	179	181	178	182
Beef	**3,859**	**3,817**	**3,927**	**3,965**	**4,039**	**4,160**	**4,117**	**4,015**	**3,942**	**3,940**	**3,869**	**3,825**	**3,821**	**3,832**	**3,730**	**3,772**
Cows	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457	2,457
Replacements 7-11 mo.	52	54	57	60	62	61	60	56	54	53	53	54	54	53	54	56
Replacements 12-23 mo.	190	196	203	216	229	232	225	216	206	198	198	200	200	201	198	204
Steer Stockers	431	403	465	483	436	480	459	430	418	400	362	352	355	361	325	337
Heifer	232	220	234	241	232	250	241	241	236	229	207	203	205	210	188	199
Feedlot Cattle	412	396	383	352	373	382	368	374	378	420	426	408	421	429	399	406
Bulls	116	116	118	119	122	127	127	123	118	119	116	116	115	115	113	114
Total	**5,234**	**5,195**	**5,302**	**5,280**	**5,353**	**5,480**	**5,372**	**5,270**	**5,192**	**5,204**	**5,153**	**5,105**	**5,110**	**5,131**	**5,014**	**5,091**

Source: EPA 2007.

Note: Totals may not sum due to independent rounding.

Appendix Table A-12 IPCC Emission Factors for Livestock

	Emission Factors
Animal Type	*(kg CH_4/head/year)*
Bulls	100
Calves	0
Swine	1.5
Sheep	8
Goats	5
Horses	18

Source: EPA 2007, IPCC 2000.

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

	CH_4	N_2O	***Total***
State		*Tg* $C0_2$ *eq.*	
Alabama	0.37	0.45	**0.82**
Alaska	0.01	0.04	**0.04**
Arizona	0.92	0.17	**1.09**
Arkansas	0.32	0.01	**0.32**
California	6.49	0.74	**7.23**
Colorado	0.60	0.52	**1.12**
Connecticut	0.03	0.01	**0.04**
Delaware	0.03	0.01	**0.04**
Florida	0.58	0.03	**0.61**
Georgia	0.67	0.02	**0.69**
Hawaii	0.05	0.05	**0.10**
Idaho	1.75	0.28	**2.03**
Illinois	1.18	0.17	**1.36**
Indiana	0.98	0.13	**1.12**
Iowa	5.83	0.57	**6.40**
Kansas	0.86	1.11	**1.97**
Kentucky	0.26	0.02	**0.28**
Louisiana	0.11	0.01	**0.13**
Maine	0.03	0.01	**0.04**
Maryland	0.09	0.02	**0.12**
Massachusetts	0.01	0.01	**0.03**
Michigan	0.73	0.19	**0.93**
Minnesota	1.96	0.38	**2.34**
Mississippi	0.47	0.02	**0.50**
Missouri	0.91	0.13	**1.04**
Montana	0.13	0.04	**0.18**
Nebraska	0.91	1.09	**2.00**
Nevada	0.12	0.02	**0.14**
New Hampshire	0.01	0.00	**0.02**
New Jersey	0.01	0.00	**0.02**
New Mexico	1.37	0.12	**1.49**
New York	0.52	0.14	**0.66**
North Carolina	4.45	0.05	**4.51**
North Dakota	0.11	0.08	**0.18**
Ohio	0.66	0.19	**0.85**
Oklahoma	1.46	0.19	**1.64**
Oregon	0.40	0.08	**0.47**
Pennsylvania	0.61	0.16	**0.76**
Rhode Island	0.00	0.01	**0.02**
South Carolina	0.30	0.00	**0.30**
South Dakota	0.53	0.22	**0.75**
Tennessee	0.17	0.01	**0.18**
Texas	2.20	1.24	**3.44**
Utah	0.50	0.08	**0.58**
Vermont	0.11	0.03	**0.14**
Virginia	0.31	0.03	**0.34**
Washington	0.75	0.16	**0.91**
West Viginia	0.03	0.01	**0.04**
Wisconsin	1.41	0.56	**1.97**
Wyoming	0.06	0.04	**0.10**
Total	**42.39**	**9.65**	**52.04**

Source: EPA 2007

[1] Methane totals include emissions from grazed land manure.

Appendix Table A-14 Methane Emissions from Manure Management by State and Animal in 2005

	Dairy cattle	Beef cattle	Poultry	Swine	Goats	Horses	Sheep	*Total*
State				*Tg* CO_2 *eq.*				
Alabama	0.0009	0.0023	0.0114	0.0029	0.0000	0.0005	0.0000	**0.0180**
Alaska	0.0001	0.0000	0.0002	0.0000	0.0000	0.0000	0.0000	**0.0003**
Arizona	0.0394	0.0014	0.0007	0.0026	0.0000	0.0004	0.0001	**0.0444**
Arkansas	0.0010	0.0032	0.0053	0.0055	0.0000	0.0006	0.0000	**0.0156**
California	0.2976	0.0042	0.0049	0.0024	0.0000	0.0010	0.0005	**0.3107**
Colorado	0.0145	0.0028	0.0032	0.0081	0.0000	0.0006	0.0002	**0.0294**
Connecticut	0.0009	0.0000	0.0006	0.0000	0.0000	0.0000	0.0000	**0.0016**
Deleware	0.0004	0.0000	0.0010	0.0002	0.0000	0.0000	0.0000	**0.0016**
Florida	0.0165	0.0028	0.0080	0.0001	0.0000	0.0008	0.0000	**0.0282**
Georgia	0.0056	0.0019	0.0197	0.0049	0.0000	0.0006	0.0000	**0.0327**
Hawaii	0.0014	0.0003	0.0002	0.0003	0.0000	0.0000	0.0000	**0.0023**
Idaho	0.0804	0.0019	0.0008	0.0001	0.0000	0.0004	0.0001	**0.0839**
Illinois	0.0066	0.0014	0.0006	0.0477	0.0000	0.0003	0.0000	**0.0566**
Indiana	0.0099	0.0007	0.0015	0.0347	0.0000	0.0005	0.0000	**0.0473**
Iowa	0.0107	0.0038	0.0021	0.2609	0.0000	0.0004	0.0001	**0.2780**
Kansas	0.0117	0.0073	0.0001	0.0219	0.0000	0.0004	0.0001	**0.0415**
Kentucky	0.0018	0.0025	0.0015	0.0064	0.0000	0.0008	0.0000	**0.0131**
Louisiana	0.0014	0.0015	0.0023	0.0001	0.0000	0.0004	0.0000	**0.0057**
Maine	0.0011	0.0000	0.0004	0.0000	0.0000	0.0001	0.0000	**0.0016**
Maryland	0.0027	0.0001	0.0012	0.0004	0.0000	0.0001	0.0000	**0.0046**
Massachusetts	0.0005	0.0000	0.0000	0.0001	0.0000	0.0001	0.0000	**0.0007**
Michigan	0.0238	0.0006	0.0009	0.0097	0.0000	0.0005	0.0000	**0.0355**
Minnesota	0.0229	0.0017	0.0018	0.0667	0.0000	0.0005	0.0001	**0.0937**
Mississippi	0.0010	0.0018	0.0113	0.0084	0.0000	0.0005	0.0000	**0.0231**
Missouri	0.0050	0.0048	0.0011	0.0325	0.0000	0.0007	0.0000	**0.0442**
Montana	0.0016	0.0029	0.0003	0.0015	0.0000	0.0005	0.0002	**0.0070**
Nebraska	0.0050	0.0075	0.0008	0.0303	0.0000	0.0003	0.0001	**0.0439**
Nevada	0.0050	0.0007	0.0000	0.0000	0.0000	0.0001	0.0000	**0.0059**
New Hampshire	0.0005	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	**0.0006**
New Jersey	0.0004	0.0000	0.0002	0.0001	0.0000	0.0001	0.0000	**0.0008**
New Mexico	0.0629	0.0015	0.0006	0.0000	0.0000	0.0002	0.0001	**0.0654**
New York	0.0233	0.0004	0.0005	0.0007	0.0000	0.0004	0.0000	**0.0254**
North Carolina	0.0017	0.0009	0.0133	0.1961	0.0000	0.0005	0.0000	**0.2126**
North Dakota	0.0015	0.0020	0.0001	0.0015	0.0000	0.0002	0.0001	**0.0054**
Ohio	0.0132	0.0010	0.0012	0.0159	0.0000	0.0007	0.0001	**0.0322**
Oklahoma	0.0104	0.0059	0.0046	0.0484	0.0000	0.0012	0.0001	**0.0707**
Oregon	0.0154	0.0019	0.0014	0.0001	0.0000	0.0005	0.0001	**0.0194**
Pennsylvania	0.0141	0.0007	0.0015	0.0125	0.0000	0.0006	0.0001	**0.0295**
Rhode Island	0.0000	0.0000	0.0001	0.0000	0.0000	0.0000	0.0000	**0.0001**
South Carolina	0.0008	0.0007	0.0061	0.0065	0.0000	0.0003	0.0000	**0.0145**
South Dakota	0.0054	0.0042	0.0003	0.0152	0.0000	0.0004	0.0002	**0.0258**
Tennessee	0.0017	0.0024	0.0008	0.0032	0.0000	0.0008	0.0000	**0.0089**
Texas	0.0555	0.0235	0.0067	0.0192	0.0004	0.0029	0.0008	**0.1090**
Utah	0.0138	0.0011	0.0029	0.0060	0.0000	0.0003	0.0001	**0.0243**
Vermont	0.0050	0.0001	0.0000	0.0000	0.0000	0.0001	0.0000	**0.0052**
Virginia	0.0024	0.0017	0.0019	0.0089	0.0000	0.0004	0.0000	**0.0153**
Washington	0.0332	0.0010	0.0012	0.0002	0.0000	0.0004	0.0000	**0.0361**
West Virginia	0.0004	0.0005	0.0005	0.0001	0.0000	0.0002	0.0000	**0.0016**
Wisconsin	0.0613	0.0013	0.0005	0.0041	0.0000	0.0005	0.0000	**0.0678**
Wyoming	0.0003	0.0016	0.0000	0.0011	0.0000	0.0003	0.0002	**0.0036**
Total	**0.8929**	**0.1108**	**0.1265**	**0.8882**	**0.0009**	**0.0223**	**0.0037**	**2.0454**

Source: EPA 2007

Appendix Table A-15 Nitrous Oxide Emissions from Manure Management by State and Animal in 2005

	Beef cattle	Dairy cattle	Poultry	Swine	*Total*
State		*Tg CO_2 eq.*			
Alabama	0.002	0.002	0.444	0.001	**0.448**
Alaska	0.000	0.000	0.035	0.000	**0.036**
Arizona	0.139	0.030	0.000	0.001	**0.170**
Arkansas	0.004	0.003	0.000	0.002	**0.009**
California	0.225	0.459	0.052	0.001	**0.737**
Colorado	0.462	0.033	0.012	0.008	**0.515**
Connecticut	0.000	0.005	0.001	0.000	**0.006**
Deleware	0.000	0.001	0.004	0.000	**0.005**
Florida	0.001	0.020	0.009	0.000	**0.030**
Georgia	0.002	0.009	0.004	0.001	**0.017**
Hawaii	0.000	0.001	0.045	0.000	**0.047**
Idaho	0.126	0.150	0.000	0.000	**0.276**
Illinois	0.088	0.042	0.000	0.044	**0.174**
Indiana	0.053	0.044	0.005	0.033	**0.135**
Iowa	0.387	0.071	0.014	0.101	**0.573**
Kansas	1.034	0.037	0.019	0.020	**1.109**
Kentucky	0.004	0.011	0.001	0.002	**0.019**
Louisiana	0.001	0.003	0.011	0.000	**0.014**
Maine	0.000	0.008	0.003	0.000	**0.011**
Maryland	0.005	0.015	0.001	0.000	**0.021**
Massachusetts	0.000	0.004	0.009	0.000	**0.013**
Michigan	0.080	0.103	0.000	0.009	**0.192**
Minnesota	0.122	0.189	0.006	0.066	**0.383**
Mississippi	0.002	0.003	0.018	0.002	**0.025**
Missouri	0.029	0.044	0.028	0.029	**0.130**
Montana	0.025	0.006	0.011	0.002	**0.044**
Nebraska	1.038	0.019	0.000	0.030	**1.086**
Nevada	0.004	0.007	0.004	0.000	**0.015**
New Hampshire	0.000	0.004	0.000	0.000	**0.004**
New Jersey	0.000	0.003	0.000	0.000	**0.003**
New Mexico	0.053	0.067	0.000	0.000	**0.120**
New York	0.010	0.130	0.000	0.001	**0.141**
North Carolina	0.002	0.007	0.001	0.044	**0.054**
North Dakota	0.025	0.013	0.036	0.002	**0.076**
Ohio	0.084	0.090	0.001	0.015	**0.190**
Oklahoma	0.149	0.016	0.011	0.012	**0.188**
Oregon	0.034	0.034	0.009	0.000	**0.077**
Pennsylvania	0.032	0.112	0.003	0.011	**0.158**
Rhode Island	0.000	0.000	0.015	0.000	**0.015**
South Carolina	0.001	0.002	0.000	0.002	**0.004**
South Dakota	0.168	0.030	0.010	0.015	**0.223**
Tennessee	0.002	0.009	0.002	0.001	**0.014**
Texas	1.143	0.088	0.007	0.004	**1.242**
Utah	0.015	0.031	0.026	0.007	**0.079**
Vermont	0.000	0.027	0.001	0.000	**0.029**
Virginia	0.013	0.012	0.000	0.002	**0.026**
Washington	0.082	0.061	0.015	0.000	**0.159**
West Virginia	0.003	0.002	0.004	0.000	**0.009**
Wisconsin	0.095	0.461	0.004	0.004	**0.563**
Wyoming	0.034	0.002	0.004	0.001	**0.040**
Total	**5.776**	**2.516**	**0.887**	**0.475**	**9.655**

Source: EPA 2007

Appendix Table A-16 Waste Characteristics Data

	Average TAM[1]	Source	Nitrogen, N_{ex}[2]	Source	Max Methane Generation Potential	Source	Volatile Solids (VS)	Source
Livestock	*kg*		*kg/day per 1,000 kg mass*		*B_o (m3 CH4/kg VS added)*		*kg/day per 1,000 kg mass*	
Dairy Cows	604	Safley 2000	0.44	USDA 1996a	0.24	Morris 1976	8.8	Lieberman and Pape, 2005
Dairy Heifers	476	Safley 2000	0.31	USDA 1996a	0.17	Bryant et. al. 1976	6.7	Lieberman and Pape, 2005
Feedlot Steers	420	USDA 1996a	0.3	USDA 1996a	0.33	Hashimoto 1981	3.86	Lieberman and Pape, 2005
Feedlot Heifers	420	USDA 1996a	0.3	USDA 1996a	0.33	Hashimoto 1981	3.98	Lieberman and Pape, 2005
Bulls NOF[3]	750	Safley 2000	0.31	USDA 1996a	0.17	Hashimoto 1981	6.04	USDA 1996a
Calves NOF	118	ERG 2003	0.3	USDA 1996a	0.17	Hashimoto 1981	6.41	USDA 1996a
Heifers NOF	420	USDA 1996a	0.31	USDA 1996a	0.17	Hashimoto 1981	7.09	Lieberman and Pape, 2005
Steers NOF	318	Safley 2000	0.31	USDA 1996a	0.17	Hashimoto 1981	7.93	Lieberman and Pape, 2005
Cows NOF	533	NRC 2000	0.33	USDA 1996a	0.17	Hashimoto 1981	6.97	Lieberman and Pape, 2005
Market Swine <60 lbs.	16	Safley 2000	0.6	USDA 1996a	0.48	Hashimoto 1984	8.8	USDA 1996a
Market Swine 60-119 lbs.	41	Safley 2000	0.42	USDA 1996a	0.48	Hashimoto 1984	5.4	USDA 1996a
Market Swine 120-179 lbs.	68	Safley 2000	0.42	USDA 1996a	0.48	Hashimoto 1984	5.4	USDA 1996a
Market Swine >180 lbs.	91	Safley 2000	0.42	USDA 1996a	0.48	Hashimoto 1984	5.4	USDA 1996a
Breeding Swine	198	Safley 2000	0.24	USDA 1996a	0.48	Hashimoto 1984	2.6	USDA 1996a
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 1996a	0.39	Hill 1982	10.8	USDA 1996a
Pullets	1.8	ASAE 1999	0.62	USDA 1996a	0.39	Hill 1982	9.7	USDA 1996a
Other Chickens	1.8	ASAE 1999	0.83	USDA 1996a	0.39	Hill 1982	10.8	USDA 1996a
Broilers	0.9	ASAE 1999	1.1	USDA 1996a	0.36	Hill 1984	15	USDA 1996a
Turkeys	6.8	ASAE 1999	0.74	USDA 1996a	0.36	Hill 1984	9.7	USDA 1996a

Source: EPA 2007.

[1] (TAM) Typical animal mass.

[2] Nitrogen excretion source.

[3] (NOF) Not on feed.

Appendix Table A-17 State Volatile Solids Production Rates in 2005

	Dairy Cow	Dairy Heifer	Cow NOF[1]	Heifer NOF	Steer NOF	Feedlot Heifer	Feedlot Steer
State				*kg/day/1000 kg mass*			
Alabama	8.76	6.81	6.74	7.21	7.76	3.91	3.78
Alaska	11.03	6.81	8.71	9.47	10.27	3.90	3.77
Arizona	11.03	6.81	8.71	9.53	10.27	3.90	3.77
Arkansas	9.19	7.56	6.72	7.19	7.74	3.94	3.81
California	9.47	6.81	6.57	7.06	7.55	3.89	3.76
Colorado	8.97	6.81	6.19	6.66	7.08	3.92	3.78
Connecticut	8.62	6.13	6.62	7.09	7.62	3.89	3.76
Delaware	8.62	6.13	6.62	7.13	7.62	3.89	3.76
Florida	8.76	6.81	6.74	7.19	7.76	3.91	3.78
Georgia	8.76	6.81	6.74	7.22	7.76	3.91	3.78
Hawaii	11.03	6.81	8.71	9.49	10.27	3.90	3.77
Idaho	11.03	6.81	8.71	9.58	10.27	3.90	3.77
Illinois	8.74	6.81	6.63	7.14	7.62	3.92	3.79
Indiana	8.74	6.81	6.63	7.13	7.62	3.92	3.79
Iowa	8.74	6.81	6.63	7.16	7.62	3.92	3.79
Kansas	8.97	6.81	6.19	6.67	7.08	3.92	3.78
Kentucky	8.76	6.81	6.74	7.23	7.76	3.91	3.78
Louisiana	9.19	7.56	6.72	7.18	7.74	3.94	3.81
Maine	8.62	6.13	6.62	7.08	7.62	3.89	3.76
Maryland	8.62	6.13	6.62	7.11	7.62	3.89	3.76
Massachusetts	8.62	6.13	6.62	7.07	7.62	3.89	3.76
Michigan	8.74	6.81	6.63	7.13	7.62	3.92	3.79
Minnesota	8.74	6.81	6.63	7.14	7.62	3.92	3.79
Mississippi	8.76	6.81	6.74	7.21	7.76	3.91	3.78
Missouri	8.74	6.81	6.63	7.11	7.62	3.92	3.79
Montana	8.97	6.81	6.19	6.59	7.08	3.92	3.78
Nebraska	8.97	6.81	6.19	6.66	7.08	3.92	3.78
Nevada	11.03	6.81	8.71	9.54	10.27	3.90	3.77
New Hampshire	8.62	6.13	6.62	7.08	7.62	3.89	3.76
New Jersey	8.62	6.13	6.62	7.10	7.62	3.89	3.76
New Mexico	11.03	6.81	8.71	9.55	10.27	3.90	3.77
New York	8.62	6.13	6.62	7.13	7.62	3.89	3.76
North Carolina	8.76	6.81	6.74	7.20	7.76	3.91	3.78
North Dakota	8.97	6.81	6.19	6.63	7.08	3.92	3.78
Ohio	8.74	6.81	6.63	7.11	7.62	3.92	3.79
Oklahoma	9.19	7.56	6.72	7.23	7.74	3.94	3.81
Oregon	11.03	6.81	8.71	9.54	10.27	3.90	3.77
Pennsylvania	8.62	6.13	6.62	7.12	7.62	3.89	3.76
Rhode Island	8.62	6.13	6.62	7.08	7.62	3.89	3.76
South Carolina	8.76	6.81	6.74	7.21	7.76	3.91	3.78
South Dakota	8.97	6.81	6.19	6.64	7.08	3.92	3.78
Tennessee	8.76	6.81	6.74	7.21	7.76	3.91	3.78
Texas	9.19	7.56	6.72	7.24	7.74	3.94	3.81
Utah	11.03	6.81	8.71	9.55	10.27	3.90	3.77
Vermont	8.62	6.13	6.62	7.10	7.62	3.89	3.76
Virginia	8.76	6.81	6.74	7.23	7.76	3.91	3.78
Washington	11.03	6.81	8.71	9.59	10.27	3.90	3.77
West Viginia	8.62	6.13	6.62	7.09	7.62	3.89	3.76
Wisconsin	8.74	6.81	6.63	7.12	7.62	3.92	3.79
Wyoming	8.97	6.81	6.19	6.62	7.08	3.92	3.78

Source: EPA 2007.

[1] (NOF) Not on feed.

Appendix Table A-18 State-Based Methane Conversion Factors[1] by Waste Management System in 2005

State	Liquid/Slurry and Deep Pit	Anaerobic Lagoon
	%	
Alabama	38.5	75.8
Alaska	13.8	48.3
Arizona	53.2	79.3
Arkansas	36.1	75.9
California	37.7	76.2
Coloradao	22.2	66.7
Connecticut	23.9	69.4
Delaware	29.7	73.9
Florida	52.2	77.8
Georgia	38.3	75.6
Hawaii	59.7	77.1
Idaho	23.2	68.3
Illinois	26.9	71.5
Indiana	26.0	70.6
Iowa	24.7	69.7
Kansas	31.9	74.5
Kentucky	30.4	73.2
Louisiana	46.1	77.2
Maine	19.5	63.3
Maryland	27.6	72.1
Massachusetts	23.2	68.7
Michigan	22.0	66.7
Minnesotta	22.8	67.9
Mississippi	40.1	76.1
Missouri	30.4	73.8
Montana	21.1	65.9
Nebraska	26.7	71.5
Nevada	25.7	70.5
New Hampshire	21.0	65.5
New Jersey	26.4	71.9
New Mexico	32.6	74.4
New York	21.7	66.6
North Carolina	33.7	74.6
North Dakota	21.7	66.9
Ohio	24.8	69.5
Oklahoma	36.5	76.1
Oregon	22.8	67.0
Pennsylvania	25.2	70.4
Rhode Island	24.6	70.4
South Carolina	37.8	75.8
South Dakota	24.2	69.6
Tennessee	32.6	74.2
Texas	41.6	77.0
Utah	26.2	71.1
Vermont	20.2	64.5
Virginia	27.9	72.0
Washington	23.4	67.9
West Virginia	25.3	69.8
Wisconsin	22.4	67.7
Wyoming	21.3	66.0

Source: EPA 2007, IPCC 2000.

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

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

	Methane	Nitrous Oxide
Manure Management System		
Pasture	0.0015	
Daily spread	0.005	
Solid storage	0.015	
Dry lot	0.015	
Poultry with bedding	0.015	
Poultry without bedding	0.015	
Liquid systems		0.001
Dry systems		0.02

Source: IPCC 2000.

Appendix Table A-20 State-Weighted Methane Conversion Factors for Livestock Waste Emissions 2005[1]

	Beef Feed Lot Heifer	Beef Feed Lot Steer	Dairy Cow	Dairy Heifer	Swine Market	Swine Breeding	Layer	Broiler	Turkey
State					%				
Alabama	2.0	1.5	18.3	1.9	54.4	54.2	32.6	1.5	1.5
Alaska	1.2	1.0	20.8	1.2	10.5	10.5	13.7	1.5	1.5
Arizona	1.7	1.5	61.4	1.7	51.4	50.9	46.7	1.5	1.5
Arkansas	2.0	1.5	11.4	1.9	53.7	53.7	1.5	1.5	1.5
California	2.0	1.5	51.1	1.8	46.1	46.7	10.3	1.5	1.5
Coloradao	1.1	1.0	45.4	1.1	29.7	29.6	39.3	1.5	1.5
Connecticut	1.3	1.0	14.5	1.2	15.0	15.0	5.0	1.5	1.5
Delaware	1.3	1.0	15.1	1.3	34.4	34.4	5.2	1.5	1.5
Florida	2.2	1.5	38.5	2.0	16.9	16.9	33.5	1.5	1.5
Georgia	2.0	1.5	21.5	1.9	52.0	52.0	32.3	1.5	1.5
Hawaii	2.3	1.5	64.9	2.1	48.1	48.1	20.4	1.5	1.5
Idaho	1.1	1.0	47.5	1.1	15.6	15.6	39.0	1.5	1.5
Illinois	1.2	1.0	20.2	1.2	34.6	34.7	2.9	1.5	1.5
Indiana	1.2	1.0	20.5	1.1	33.0	33.0	1.5	1.5	1.5
Iowa	1.2	1.0	18.4	1.1	46.4	46.4	1.5	1.5	1.5
Kansas	1.2	1.0	33.5	1.2	35.3	35.3	3.0	1.5	1.5
Kentucky	1.3	1.0	5.2	1.3	48.8	48.7	5.2	1.5	1.5
Louisiana	2.1	1.5	11.5	2.0	25.3	25.3	47.2	1.5	1.5
Maine	1.2	1.0	10.5	1.2	8.3	8.3	4.6	1.5	1.5
Maryland	1.3	1.0	12.1	1.3	30.6	30.6	5.2	1.5	1.5
Massachusetts	1.2	1.0	9.1	1.2	22.8	22.8	4.9	1.5	1.5
Michigan	1.1	1.0	25.0	1.1	30.7	30.7	2.9	1.5	1.5
Minnesotta	1.1	1.0	15.9	1.1	30.8	30.8	1.5	1.5	1.5
Mississippi	2.0	1.5	12.6	1.9	58.9	58.9	46.5	1.5	1.5
Missouri	1.2	1.0	13.6	1.2	35.7	35.7	1.5	1.5	1.5
Montana	1.1	1.0	27.9	1.1	23.7	23.7	37.6	1.5	1.5
Nebraska	1.2	1.0	25.9	1.1	32.3	32.2	2.9	1.5	1.5
Nevada	1.1	1.0	51.3	1.1	33.8	32.3	1.5	1.5	1.5
New Hampshire	1.2	1.0	11.0	1.2	13.9	13.9	4.8	1.5	1.5
New Jersey	1.3	1.0	10.0	1.2	26.0	26.0	5.1	1.5	1.5
New Mexico	1.1	1.0	50.9	1.1	3.9	3.9	42.7	1.5	1.5
New York	1.2	1.0	11.6	1.2	24.5	24.6	4.9	1.5	1.5
North Carolina	1.3	1.0	9.6	1.3	59.4	59.4	32.2	1.5	1.5
North Dakota	1.1	1.0	14.4	1.1	25.7	25.7	2.8	1.5	1.5
Ohio	1.2	1.0	16.0	1.1	31.3	31.3	1.5	1.5	1.5
Oklahoma	1.1	1.0	43.0	1.6	57.1	57.2	46.8	1.5	1.5
Oregon	1.3	1.0	32.8	1.2	12.7	12.7	17.1	1.5	1.5
Pennsylvania	1.3	1.0	8.0	1.2	32.9	32.9	1.5	1.5	1.5
Rhode Island	1.3	1.0	7.5	1.2	15.5	15.5	5.0	1.5	1.5
South Carolina	2.0	1.5	15.0	1.9	54.5	54.5	46.4	1.5	1.5
South Dakota	1.2	1.0	21.4	1.1	31.4	31.4	2.9	1.5	1.5
Tennessee	1.3	1.0	7.3	1.3	47.5	47.2	5.1	1.5	1.5
Texas	1.6	1.5	53.8	1.6	57.5	57.6	10.7	1.5	1.5
Utah	1.1	1.0	40.4	1.1	26.6	26.7	39.6	1.5	1.5
Vermont	1.2	1.0	11.4	1.2	5.0	5.0	4.7	1.5	1.5
Virginia	1.3	1.0	7.3	1.2	53.1	53.2	5.1	1.5	1.5
Washington	1.3	1.0	36.3	1.2	18.2	18.2	9.0	1.5	1.5
West Virginia	1.3	1.0	10.7	1.2	15.1	15.1	5.0	1.5	1.5
Wisconsin	1.1	1.0	15.9	1.1	27.9	27.9	2.9	1.5	1.5
Wyoming	1.1	1.0	23.3	1.1	25.9	25.9	38.1	1.5	1.5

Source: EPA 2007

[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	*Tg N*
1990	3.9
1991	3.9
1992	4.0
1993	4.0
1994	4.1
1995	4.2
1996	4.2
1997	4.0
1998	3.9
1999	3.9
2000	3.8
2001	3.8
2002	3.8
2003	3.8
2004	3.7
2005	3.7

Source: EPA 2007

Appendix B

B-1 Rice Harvested Area, 1990-2005
B-2 Total U.S. Production of Crops Managed with Burning, 1990-2005
B-3 State Production of Crops Managed with Burning in 2005
B-4 Information used in Estimating Methane and Nitrous Oxide Emissions from Crop Residue Burning; (a) Crop Assumptions and Coefficients; (b) Emissions Factors and Global Warming Potentials; (c) Rice Area Burned, by State
B-5 State Methane Emission Estimates for Cropland Burning by Crop Type in 2005
B-6 State Estimates of Nitrous Oxide Emissions from Cropland Burning by Crop Type in 2005
B-7 Soil Carbon Stocks by Climate Region and U.S. Soil Groupings
B-8 Stock Change Factors for the U.S. and IPCC Default Values for Impacts on Mineral Soils
B-9 Cultivated Histosol (Organic Soils) Area
B-10 Carbon Loss Rates from Organic Soils Under Agricultural Management in the United States
B-11 Areas in Each Land-Use and Management System, for all U.S. Land Areas Categorized as in Agricultural Use
B-12 State-Level Estimates of Annual Soil Carbon Stock Changes by Major Land Use and Management Type, 1997
B-13 State-Level Estimates of Soil Carbon Changes on Cropland by Major Activity, 1997

Appendix Table B-1 Rice Harvested Area, 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
State and Crop								*1,000 Hectares*								
Arkansas	**486**	**510**	**558**	**498**	**575**	**542**	**473**	**563**	**601**	**658**	**571**	**656**	**608**	**589**	**629**	**662**
Primary	486	510	558	498	575	542	473	563	601	658	571	656	608	589	629	662
Ratoon	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
California	**160**	**144**	**159**	**177**	**196**	**188**	**202**	**209**	**185**	**204**	**222**	**191**	**214**	**205**	**239**	**213**
Florida	**7**	**13**	**14**	**14**	**15**	**15**	**13**	**12**	**12**	**12**	**11**	**7**	**8**	**5**	**8**	**7**
Primary	5	9	9	9	10	10	9	8	8	7	8	5	5	2	5	4
Ratoon	2	4	5	5	5	5	4	4	4	5	3	3	3	2	3	3
Louisiana	**287**	**268**	**326**	**279**	**326**	**300**	**280**	**307**	**326**	**324**	**272**	**287**	**249**	**246**	**280**	**240**
Primary	221	206	251	214	251	231	216	236	251	249	194	221	217	182	216	212
Ratoon	66	62	75	64	75	69	65	71	75	75	78	66	32	64	65	28
Mississippi	**101**	**89**	**111**	**99**	**127**	**117**	**84**	**96**	**108**	**131**	**88**	**102**	**102**	**95**	**95**	**106**
Missouri	**32**	**37**	**45**	**38**	**50**	**45**	**38**	**47**	**58**	**74**	**68**	**84**	**74**	**69**	**79**	**87**
Texas	**200**	**194**	**199**	**169**	**201**	**180**	**169**	**147**	**160**	**147**	**130**	**122**	**114**	**101**	**119**	**103**
Primary	143	139	142	121	143	129	121	105	115	105	87	87	83	73	88	81
Ratoon	57	56	57	48	57	51	48	42	46	42	43	35	31	28	31	22
Total	**1,273**	**1,256**	**1,414**	**1,273**	**1,489**	**1,387**	**1,261**	**1,380**	**1,452**	**1,550**	**1,362**	**1,450**	**1,369**	**1,309**	**1,449**	**1,418**

Appendix Table B-2 Total U.S. Production of Crops Managed with Burning, 1990-2005

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Crop								*Million Metric tons*								
Wheat	74.3	53.9	67.1	65.2	63.2	59.4	62.0	67.5	69.3	62.6	60.8	53.3	43.7	63.9	58.8	57.3
Rice	7.1	7.3	8.2	7.1	9.0	7.9	7.8	8.3	8.6	9.4	8.7	9.7	9.6	9.1	10.5	10.1
Sugarcane	25.5	27.4	27.5	28.2	28.1	27.9	26.7	28.8	30.9	32.0	32.8	31.6	32.3	30.7	26.3	22.5
Corn	201.5	189.9	240.7	161.0	255.3	188.0	234.5	233.9	247.9	239.5	251.9	241.5	228.0	256.5	300.2	282.6
Barley	9.2	10.1	9.9	8.7	8.2	7.8	8.5	7.8	7.7	6.1	6.9	5.4	4.9	6.1	6.1	4.6
Soybeans	52.4	54.1	59.6	50.9	68.4	59.2	64.8	73.2	74.6	72.2	75.1	78.7	75.1	66.8	85.1	83.4
Peanuts	1.6	2.2	1.9	1.5	1.9	1.6	1.7	1.6	1.8	1.7	1.5	1.9	1.5	1.9	1.9	2.2
Total	**371.7**	**344.9**	**415.1**	**322.6**	**434.1**	**351.8**	**406.0**	**421.1**	**440.7**	**423.6**	**437.5**	**422.0**	**395.1**	**435.0**	**489.0**	**462.8**

Appendix Table B-3 State Production of Crops Managed with Burning in 2005

	Corn	Soybeans	Barley	Wheat	Peanuts	Rice	Sugarcane
State	*1,000 bushels*				*1,000 lbs.*	*1,000 cwt*	*1,000 tons*
Alabama	23,800	4,785	-	2,250	613,250	-	-
Alaska	ND[1]	ND	208,000	ND	ND	ND	ND
Arizona	4,290	-	3,000	8,060	-	-	-
Arkansas	30,130	102,000	-	8,320	-	108,792	-
California	22,360	-	3,780	28,155	-	38,836	-
Colorado	140,600	-	7,670	54,035	-	-	-
Connecticut	-	-	-	-	-	-	-
Delaware	22,022	4,732	2,187	3,570	-	-	-
Florida	2,632	256	-	360	410,400	-	11,806
Georgia	29,670	4,550	-	7,280	2,130,000	-	-
Hawaii	-	-	-	-	-	-	1,753
Idaho	10,200	-	52,200	100,590	-	-	-
Illinois	1,708,850	439,425	-	36,600	-	-	-
Indiana	888,580	263,620	-	24,480	-	-	-
Iowa	2,162,500	525,000	-	750	-	-	-
Kansas	465,750	105,450	588	380,000	-	-	-
Kentucky	155,760	53,320	747	20,400	-	-	-
Louisiana	44,880	28,900	-	4,800	-	30,983	9,618
Maine	-	-	1,320	-	-	-	-
Maryland	54,000	15,980	3,526	9,240	-	-	-
Massachusetts	-	-	-	-	-	-	-
Michigan	287,430	76,615	517	38,940	-	-	-
Minnesota	1,191,900	306,000	3,870	71,470	-	-	-
Mississippi	47,085	58,035	-	3,250	44,800	16,832	-
Missouri	329,670	181,670	-	29,160	-	14,124	-
Montana	2,516	-	39,200	192,480	-	-	-
Nebraska	1,270,500	235,330	-	68,640	-	-	-
Nevada	-	-	170	805	-	-	-
New Hampshire	-	-	-	-	-	-	-
New Jersey	7,564	2,548	142	1,219	-	-	-
New Mexico	9,625	-	-	9,720	66,500	-	-
New York	57,040	7,896	735	5,130	-	-	-
North Carolina	84,000	39,420	1,482	24,795	288,000	-	-
North Dakota	154,800	104,400	57,240	303,765	-	-	-
Ohio	464,750	201,600	300	58,930	-	-	-
Oklahoma	28,750	7,930	-	128,000	107,910	-	-
Oregon	4,000	-	2,025	53,560	-	-	-
Pennsylvania	117,120	17,220	3,384	7,830	-	-	-
Rhode Island	-	-	-	-	-	-	-
South Carolina	33,060	8,610	-	8,580	168,000	-	-
South Dakota	470,050	134,750	2,303	133,420	-	-	-
Tennessee	77,350	41,800	-	8,400	-	-	-
Texas	210,900	5,980	-	96,000	975,000	13,668	1,551
Utah	1,956	-	1,920	7,099	-	-	-
Vermont	-	-	-	-	-	-	-
Virginia	42,480	15,300	3,915	10,080	66,000	-	-
Washington	16,400	-	12,505	139,300	-	-	-
West Virginia	3,052	595	-	300	-	-	-
Wisconsin	429,200	69,520	1,590	10,262	-	-	-
Wyoming	6,860	-	5,580	4,665	-	-	-
Total	**11,114,082**	**3,063,237**	**419,896**	**2,104,690**	**4,869,860**	**223,235**	**24,728**

[1] (ND) No data available.

(-) Indicates not applicable.

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

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	0.8
Fraction Residue Burned	0.03	0.03	0.03	0.03	0.03	Variable	0.03
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

GHG	Factor & GWP
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	22%
California	12%
Florida[1]	0%
Louisiana	3%
Mississippi	23%
Missouri	18%
Texas	0%

[1] Crop residue burning is illegal in Florida.

Table B-5 State Methane Emission Estimates for Cropland Burning by Crop Type in 2005

	Corn	Peanuts	Soybeans	Barley	Wheat	Rice	Sugarcane	***Total***
State				*Tg CO2 eq.*				
Alabama	0.0008	0.0004	0.0004	-	0.0001	-	-	**0.0017**
Alaska	ND	ND	ND	ND	ND	ND	ND	**ND**
Arizona	0.0002	-	-	0.0001	0.0004	-	-	**0.0007**
Arkansas	0.0011	-	0.0078	-	0.0004	0.0273	-	**0.0367**
California	0.0008	-	-	-	0.0014	0.0222	-	**0.0244**
Colorado	0.0050	-	-	-	0.0027	-	-	**0.0077**
Connecticut	ND	-	-	-	-	-	-	**0.0000**
Delaware	0.0008	-	0.0004	-	0.0002	-	-	**0.0013**
Florida	0.0001	0.0002	0.0000	-	0.0000	-	0.0077	**0.0081**
Georgia	0.0011	0.0013	0.0003	-	0.0004	-	-	**0.0030**
Hawaii	-	-	-	-	-	-	0.0011	**0.0011**
Idaho	0.0004	-	-	0.0020	0.0050	-	-	**0.0073**
Illinois	0.0606	-	0.0337	-	0.0018	-	-	**0.0962**
Indiana	0.0315	-	0.0202	-	0.0012	-	-	**0.0530**
Iowa	0.0767	-	0.0403	-	0.0000	-	-	**0.1171**
Kansas	0.0165	-	0.0081	0.0000	0.0190	-	-	**0.0436**
Kentucky	0.0055	-	0.0041	0.0000	0.0010	-	-	**0.0107**
Louisiana	0.0016	-	0.0022	-	0.0002	0.0031	0.0063	**0.0134**
Maine	-	-	-	0.0000	-	-	-	**0.0000**
Maryland	0.0019	-	0.0012	0.0001	0.0005	-	-	**0.0037**
Massachusetts	-	-	-	-	-	-	-	**0.0000**
Michigan	0.0102	-	0.0059	0.0000	0.0019	-	-	**0.0180**
Minnesota	0.0423	-	0.0235	0.0001	0.0036	-	-	**0.0695**
Mississippi	0.0017	0.0000	0.0045	-	0.0002	0.0169	-	**0.0232**
Missouri	0.0117	-	0.0139	-	0.0015	0.0018	-	**0.0289**
Montana	0.0001	-	-	0.0015	0.0096	-	-	**0.0112**
Nebraska	0.0451	-	0.0181	-	0.0034	-	-	**0.0666**
Nevada	-	-	-	0.0000	0.0000	-	-	**0.0000**
New Hampshire	-	-	-	-	-	-	-	**0.0000**
New Jersey	0.0003	-	0.0002	0.0000	0.0001	-	-	**0.0005**
New Mexico	0.0003	0.0000	-	-	0.0005	-	-	**0.0009**
New York	0.0020	-	0.0006	0.0000	0.0003	-	-	**0.0029**
North Carolina	0.0030	0.0002	0.0030	0.0001	0.0012	-	-	**0.0075**
North Dakota	0.0055	-	0.0080	0.0021	0.0152	-	-	**0.0308**
Ohio	0.0165	-	0.0155	0.0000	0.0029	-	-	**0.0349**
Oklahoma	0.0010	0.0001	0.0006	-	0.0064	-	-	**0.0081**
Oregon	0.0001	-	-	0.0001	0.0027	-	-	**0.0029**
Pennsylvania	0.0042	-	0.0013	0.0001	0.0004	-	-	**0.0060**
Rhode Island	-	-	-	-	-	-	-	**0.0000**
South Carolina	0.0012	0.0001	0.0007	-	0.0004	-	-	**0.0024**
South Dakota	0.0167	-	0.0103	0.0001	0.0067	-	-	**0.0338**
Tennessee	0.0027	-	0.0032	-	0.0004	-	-	**0.0064**
Texas	0.0075	0.0006	0.0005	-	0.0048	-	0.0010	**0.0143**
Utah	0.0001	-	-	0.0001	0.0004	0.0000	-	**0.0005**
Vermont	-	-	-	-	-	-	-	**0.0000**
Virginia	0.0015	0.0000	0.0012	0.0001	0.0005	-	-	**0.0034**
Washington	0.0006	-	-	0.0005	0.0070	-	-	**0.0080**
West Virginia	0.0001	-	0.0000	-	0.0000	-	-	**0.0002**
Wisconsin	0.0152	-	0.0053	0.0001	0.0005	-	-	**0.0211**
Wyoming	0.0002	-	-	0.0002	0.0002	-	-	**0.0007**
Total	**0.3944**	**0.0029**	**0.2350**	**0.0074**	**0.1051**	**0.0714**	**0.0162**	**0.8324**

- = Zero.

ND = No data.

Table B-6 State Estimates of Nitrous Oxide Emissions from Cropland Burning by Crop

	Corn	Peanuts	Soybeans	Barley	Wheat	Rice	Sugarcane	***Total***
State				*Tg CO2 eq.*				
Alabama	0.0003	0.0002	0.0005	-	0.0000	-	-	**0.0010**
Alaska	ND	ND	ND	0.0033	ND	ND	ND	**ND**
Arizona	0.0000	-	-	0.0000	0.0001	-	-	**0.0002**
Arkansas	0.0003	-	0.0098	-	0.0001	0.0215	-	**0.0317**
California	0.0003	-	-	0.0001	0.0005	0.0049	-	**0.0057**
Colorado	0.0016	-	-	0.0001	0.0009	-	-	**0.0026**
Connecticut	-	-	-	-	-	-	-	**0.0000**
Delaware	0.0002	-	0.0005	0.0000	0.0001	-	-	**0.0008**
Florida	0.0000	0.0001	0.0000	-	0.0000	-	0.0018	**0.0020**
Georgia	0.0003	0.0007	0.0004	-	0.0001	-	-	**0.0016**
Hawaii	-	-	-	-	-	-	0.0003	**0.0003**
Idaho	0.0001	-	-	0.0008	0.0017	-	-	**0.0026**
Illinois	0.0192	-	0.0420	-	0.0006	-	-	**0.0618**
Indiana	0.0100	-	0.0252	-	0.0004	-	-	**0.0356**
Iowa	0.0243	-	0.0502	-	0.0000	-	-	**0.0745**
Kansas	0.0052	-	0.0101	0.0000	0.0065	-	-	**0.0218**
Kentucky	0.0017	-	0.0051	0.0000	0.0003	-	-	**0.0072**
Louisiana	0.0005	-	0.0028	-	0.0001	0.0011	0.0014	**0.0059**
Maine	-	-	-	0.0000	-	-	-	**0.0000**
Maryland	0.0006	-	0.0015	0.0001	0.0002	-	-	**0.0023**
Massachusetts	-	-	-	-	-	-	-	**0.0000**
Michigan	0.0032	-	0.0073	0.0000	0.0007	-	-	**0.0112**
Minnesota	0.0134	-	0.0293	0.0001	0.0012	-	-	**0.0439**
Mississippi	0.0005	0.0000	0.0056	-	0.0001	0.0055	-	**0.0116**
Missouri	0.0037	-	0.0174	-	0.0005	0.0007	-	**0.0222**
Montana	0.0000	-	-	0.0006	0.0033	-	-	**0.0039**
Nebraska	0.0142	-	0.0225	-	0.0012	-	-	**0.0379**
Nevada	-	-	-	0.0000	0.0000	-	-	**0.0000**
New Hampshire	-	-	-	-	-	-	-	**0.0000**
New Jersey	0.0001	-	0.0002	0.0000	0.0000	-	-	**0.0004**
New Mexico	0.0001	0.0000	-	-	0.0002	-	-	**0.0003**
New York	0.0006	-	0.0008	0.0000	0.0001	-	-	**0.0015**
North Carolina	0.0009	0.0001	0.0038	0.0000	0.0004	-	-	**0.0053**
North Dakota	0.0017	-	0.0100	0.0009	0.0052	-	-	**0.0178**
Ohio	0.0052	-	0.0193	0.0000	0.0010	-	-	**0.0255**
Oklahoma	0.0003	0.0000	0.0008	-	0.0022	-	-	**0.0033**
Oregon	0.0000	-	-	0.0000	0.0009	-	-	**0.0010**
Pennsylvania	0.0013	-	0.0016	0.0001	0.0001	-	-	**0.0031**
Rhode Island	-	-	-	-	-	-	-	**0.0000**
South Carolina	0.0004	0.0001	0.0008	-	0.0001	-	-	**0.0014**
South Dakota	0.0053	-	0.0129	0.0000	0.0023	-	-	**0.0205**
Tennessee	0.0009	-	0.0040	-	0.0001	-	-	**0.0050**
Texas	0.0024	0.0003	0.0006	-	0.0016	0.0000	0.0002	**0.0051**
Utah	0.0000	-	-	0.0000	0.0001	-	-	**0.0002**
Vermont	-	-	-	-	-	-	-	**0.0000**
Virginia	0.0005	0.0000	0.0015	0.0001	0.0002	-	-	**0.0022**
Washington	0.0002	-	-	0.0002	0.0024	-	-	**0.0028**
West Virginia	0.0000	-	0.0001	-	0.0000	-	-	**0.0001**
Wisconsin	0.0048	-	0.0067	0.0000	0.0002	-	-	**0.0117**
Wyoming	0.0001	-	-	0.0001	0.0001	-	-	**0.0002**
Total	**0.1246**	**0.0017**	**0.2930**	**0.0066**	**0.0359**	**0.0336**	**0.0037**	**0.4991**

- = Zero.

ND = No data.

Appendix Table B-7 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**			*Tg C/ha*			
High Clay Activity Mineral Soils	Vertisols, Mollisols, Inceptisols, Aridisols, & High Base Status	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

[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.

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); sample size provided in parentheses.

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-8 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
General Uncultivated	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	na	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	1.01 ± 0.03	1.08 ± 0.03	1.01 ± 0.03
No-till	1.1	1.13 ± 0.02	1.05 ± 0.03	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

Appendix Table B-9 Cultivated Histosol (Organic Soils) Area

	Temperate	Sub-Tropical
Year	*1,000 ha*	
1990	432	192
1991	431	193
1992	429	194
1993	431	194
1994	433	195
1995	435	195
1996	437	196
1997	439	196
1998	441	197
1999	443	197
2000	445	197
2001	447	198
2002	449	198
2003	451	199
2004	453	199
2005	455	199

Appendix Table B-10 Carbon Loss Rates from Organic Soils Under Agricultural Management in the United States

	Cropland	Pasture/Forest[1]
Climate Regions	*Tg 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

[1] There is not enough data available to estimate values for C losses from managed pastures and forests. Estimates are 25% of the values for cropland (the IPCC default organic soil C losses on pasture/forest lands).

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-11 Areas in Each Land-Use and Management System, for all U.S. Land Areas Categorized as in Agricultural Use

	1982	1992	1997
IPCC Land Use/Management Categories		*million hectares*	
Medium Input Cropping	87.49	77.17	78.27
High Input Cropping (Hay or Legumes in Rotation, Winter Cover Crop, Irrigated)	22.21	22.02	21.74
Low Input Cropping (Fallow, Low Residue Crops)	30.96	28.92	25.13
Rice	2.71	2.13	2.22
Uncultivated Land (Hay land, Rangeland, Pasture, Forest, Federal)	210.04	207.77	210.26
Improved Land (Pasture or Hayland with legumes or irrigation, Continuous Perennial Crops)	31.19	33.65	31.43
CRP (Set-Aside)	0	13.78	13.23
Urban, Water, Miscellaneous Non-Cropland	1.78	0.96	4.11
Total	**386.39**	**386.39**	**386.39**

Note: Based on analysis of the Revised 1997 NRI data (NRCS 2000).

Appendix Table B-12 State-Level Estimates of Annual Soil Carbon Stock Changes by Major Land Use and Management Type, 1997

	Net Change, Cropland[1]	Net Change, Hay/Grazing Land	CRP	Manure Application	Total, Non-Organic Soils	Ag. Land on Organic Soils	***Total*** [2]
State				*Tg CO2 eq.*			
Alabama	(0.22)	0.88	0.29	0.53	1.49	(0.04)	**1.45**
Alaska	ND	ND	ND	ND	ND	ND	**ND**
Arizona	(0.18)	0.18	0.04	0.19	0.23	0.00	**0.23**
Arkansas	0.22	0.55	0.15	0.88	1.80	0.00	**1.80**
California	(0.62)	0.99	0.11	1.72	2.20	(2.35)	**(0.15)**
Colorado	(0.62)	0.84	1.25	0.53	2.00	0.00	**2.00**
Connecticut	(0.04)	0.07	0.00	0.04	0.07	(0.04)	**0.04**
Delaware	0.00	0.00	0.00	0.15	0.15	0.00	**0.15**
Florida	(0.07)	0.44	0.07	0.23	0.67	(10.74)	**(10.07)**
Georgia	(0.18)	0.73	0.18	0.76	1.50	(0.07)	**1.42**
Hawaii	0.04	0.04	0.00	n.d.	0.07	(0.29)	**(0.22)**
Idaho	(1.03)	1.36	0.59	0.34	1.26	(0.07)	**1.19**
Illinois	(2.57)	2.05	0.59	0.53	0.61	(0.84)	**(0.24)**
Indiana	(1.03)	1.36	0.26	0.61	1.20	(1.98)	**(0.78)**
Iowa	(4.33)	2.46	0.77	1.49	0.39	(1.87)	**(1.48)**
Kansas	(1.06)	2.02	1.54	0.88	3.37	0.00	**3.37**
Kentucky	(0.77)	1.65	0.11	0.15	1.14	0.00	**1.14**
Louisiana	0.11	0.51	0.11	0.04	0.77	(0.07)	**0.70**
Maine	(0.11)	0.22	0.00	0.08	0.19	0.00	**0.19**
Maryland	(0.15)	0.22	0.00	0.08	0.15	0.00	**0.15**
Massachusetts	(0.07)	0.11	0.00	0.19	0.23	(0.07)	**0.15**
Michigan	(1.94)	2.31	0.15	0.46	0.97	(3.12)	**(2.14)**
Minnesota	(4.58)	3.63	0.95	1.18	1.18	(5.24)	**(4.06)**
Mississippi	0.00	0.84	0.40	0.42	1.67	(0.04)	**1.63**
Missouri	(1.21)	1.72	0.70	0.65	1.86	(0.04)	**1.82**
Montana	(1.32)	1.83	1.80	0.08	2.39	(0.11)	**2.28**
Nebraska	(2.42)	2.42	0.99	1.03	2.02	0.00	**2.02**
Nevada	(0.11)	0.26	0.00	0.04	0.18	0.00	**0.18**
New Hampshire	(0.04)	0.00	0.00	0.00	(0.04)	(0.04)	**(0.07)**
New Jersey	(0.07)	0.15	0.00	0.04	0.11	(0.04)	**0.07**
New Mexico	(0.26)	0.40	0.22	0.23	0.60	0.00	**0.60**
New York	(1.54)	2.60	0.00	0.61	1.67	(0.48)	**1.20**
North Carolina	(0.11)	0.55	0.11	1.26	1.81	(1.06)	**0.75**
North Dakota	(1.87)	2.57	2.71	0.11	3.52	(0.22)	**3.30**
Ohio	(1.43)	1.91	0.22	0.53	1.23	(1.25)	**(0.02)**
Oklahoma	(0.44)	1.47	0.77	0.46	2.26	0.00	**2.26**
Oregon	(0.26)	0.70	0.29	0.15	0.89	(0.18)	**0.70**
Pennsylvania	(1.10)	1.65	0.00	0.80	1.35	(0.07)	**1.28**
Rhode Island	0.00	0.00	0.00	0.00	0.00	0.00	**0.00**
South Carolina	(0.11)	0.29	0.07	0.19	0.45	(0.62)	**(0.18)**
South Dakota	(3.89)	3.30	1.39	0.31	1.11	(0.07)	**1.04**
Tennessee	(0.44)	1.54	0.11	0.15	1.36	0.00	**1.36**
Texas	(2.05)	3.74	2.13	1.53	5.34	0.00	**5.34**
Utah	(0.29)	0.73	0.15	0.15	0.74	0.00	**0.74**
Vermont	(0.07)	0.15	0.00	0.11	0.19	0.00	**0.19**
Virginia	(0.29)	0.77	0.07	0.34	0.89	(0.40)	**0.49**
Washington	(0.26)	0.81	0.81	0.27	1.62	(0.22)	**1.40**
West Virginia	(0.26)	0.51	0.00	0.08	0.33	0.00	**0.33**
Wisconsin	(4.84)	4.36	0.15	1.30	0.97	(2.93)	**(1.96)**
Wyoming	(0.44)	0.95	0.37	0.04	0.92	0.00	**0.92**
Total	**(44.33)**	**58.85**	**20.61**	**21.97**	**57.10**	**(34.58)**	**22.52**

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

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

ND= No data.

Appendix Table B-13 State-Level Estimates of Soil Carbon Changes on Cropland, 1997

	Grassland Converted to Annual Cropland[1]	Management Changes	Changes on Other Cropland[2]	***Net Total***
State		*Tg CO_2 Eq.*		
Alabama	(0.37)	0.11	0.04	**(0.22)**
Alaska	ND	ND	ND	**ND**
Arizona	(0.22)	0.00	0.04	**(0.18)**
Arkansas	(0.81)	0.22	0.81	**0.22**
California	(1.14)	0.04	0.48	**(0.62)**
Colorado	(0.77)	0.15	0.00	**(0.62)**
Connecticut	(0.04)	0.00	0.00	**(0.04)**
Delaware	(0.04)	0.04	0.00	**0.00**
Florida	(0.33)	0.04	0.22	**(0.07)**
Georgia	(0.29)	0.07	0.04	**(0.18)**
Hawaii	0.00	0.00	0.04	**0.04**
Idaho	(1.10)	0.07	0.00	**(1.03)**
Illinois	(3.08)	0.48	0.04	**(2.57)**
Indiana	(1.61)	0.55	0.04	**(1.03)**
Iowa	(4.44)	0.11	0.00	**(4.33)**
Kansas	(2.05)	0.99	0.00	**(1.06)**
Kentucky	(0.95)	0.11	0.07	**(0.77)**
Louisiana	(1.14)	0.22	1.03	**0.11**
Maine	(0.11)	0.00	0.00	**(0.11)**
Maryland	(0.18)	0.04	0.00	**(0.15)**
Massachusetts	(0.07)	0.00	0.00	**(0.07)**
Michigan	(2.09)	0.07	0.07	**(1.94)**
Minnesota	(4.62)	0.00	0.04	**(4.58)**
Mississippi	(0.88)	0.22	0.66	**0.00**
Missouri	(1.91)	0.44	0.26	**(1.21)**
Montana	(1.91)	0.59	0.00	**(1.32)**
Nebraska	(3.08)	0.66	0.00	**(2.42)**
Nevada	(0.11)	0.00	0.00	**(0.11)**
New Hampshire	(0.04)	0.00	0.00	**(0.04)**
New Jersey	(0.11)	0.04	0.00	**(0.07)**
New Mexico	(0.26)	0.00	0.00	**(0.26)**
New York	(1.61)	0.04	0.04	**(1.54)**
North Carolina	(0.22)	0.07	0.04	**(0.11)**
North Dakota	(3.15)	1.28	0.00	**(1.87)**
Ohio	(1.87)	0.40	0.04	**(1.43)**
Oklahoma	(0.88)	0.44	0.00	**(0.44)**
Oregon	(0.33)	0.04	0.04	**(0.26)**
Pennsylvania	(1.21)	0.07	0.04	**(1.10)**
Rhode Island	0.00	0.00	0.00	**0.00**
South Carolina	(0.15)	0.04	0.00	**(0.11)**
South Dakota	(4.07)	0.18	0.00	**(3.89)**
Tennessee	(0.70)	0.07	0.18	**(0.44)**
Texas	(3.01)	0.59	0.37	**(2.05)**
Utah	(0.29)	0.00	0.00	**(0.29)**
Vermont	(0.07)	0.00	0.00	**(0.07)**
Virginia	(0.37)	0.07	0.00	**(0.29)**
Washington	(0.51)	0.15	0.11	**(0.26)**
West Virginia	(0.26)	0.00	0.00	**(0.26)**
Wisconsin	(4.84)	0.00	0.00	**(4.84)**
Wyoming	(0.51)	0.07	0.00	**(0.44)**
Total	**(57.82)**	**8.84**	**4.69**	**(44.29)**

[1] Losses from annual cropping systems due to plowing of pastures, rangeland, hayland, set-aside lands, and perennial cropland.

[2] Perennial/horticultural cropland and rice cultivation.

ND= No data.

Appendix C: Carbon Stocks

Appendix C

C-1 Forest Area, Stock, and Stock Change by State
C-2 Carbon Stock Pools on Private Forestland by Region and Age-Class
C-3 Carbon Stock Pools on Public Forestland by Region and Age-Class
C-4 Carbon Stock Pools on Timberlands by Region and Stand Size Class
C-5 Carbon Stocks on all Forestland by Forest Type Group and Ownership
C-6 Net Annual Carbon Stock Change on all Forestland by Forest Type Group and Ownership

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	9,286	(4.7)	2,435	1467	(11.1)	(9.6)
Arizona	8,395	60.0	1,647	712	2.4	(0.1)
Arkansas	7,620	2.1	2,356	1173	(13.5)	(4.7)
California	13,451	56.9	7,776	1878	(75.4)	(4.1)
Colorado	9,425	60.6	3,382	1087	(23.9)	(0.1)
Connecticut	704	(8.5)	326	158	3.2	(0.1)
Delaware	159	0.7	70	37	(1.2)	(0.1)
Florida	6,534	12.4	1,597	2435	(20.6)	(3.9)
Georgia	10,006	27.2	2,760	3023	(31.5)	(9.7)
Idaho	8,963	5.1	3,833	1348	7.1	(1.8)
Illinois	1,790	6.7	732	368	(2.6)	(0.3)
Indiana	1,913	34.0	829	385	(24.3)	(0.5)
Iowa	1,112	38.0	404	244	(14.1)	(0.1)
Kansas	860	26.8	270	258	(6.6)	(0.0)
Kentucky	4,844	(19.2)	1,734	719	(3.3)	(1.4)
Louisiana	5,713	11.8	1,637	964	(0.2)	(4.6)
Maine	7,165	0.3	2,666	2174	2.9	(3.5)
Maryland	953	(16.7)	449	224	1.5	(0.3)
Massachusetts	1,282	2.6	659	324	(10.0)	(0.1)
Michigan	7,815	1.3	2,945	4292	6.8	(2.6)
Minnesota	6,554	(18.3)	1,905	4054	5.9	(2.2)
Mississippi	7,525	93.5	1,933	1202	(9.4)	(7.6)
Missouri	5,933	36.6	2,084	1054	(38.0)	(1.0)
Montana	10,446	80.2	4,177	1506	(23.7)	(1.2)
Nebraska	507	14.4	169	133	(4.5)	(0.1)
Nevada	4,807	39.3	927	380	(4.8)	(0.0)
New Hampshire	1,948	(0.7)	920	529	(2.5)	(1.1)
New Jersey	776	(13.8)	324	191	(0.9)	(0.0)
New Mexico	6,751	22.8	1,759	594	(14.4)	(0.1)
New York	7,472	10.4	3,323	1960	(20.9)	(1.1)
North Carolina	7,393	(20.4)	2,503	1866	(0.3)	(6.1)
North Dakota	293	2.4	86	88	(1.1)	(0.0)
Ohio	3,281	8.5	1,286	768	(13.9)	(0.7)
Oklahoma	3,102	37.2	825	465	(16.4)	(0.8)
Oregon	12,332	26.3	7,059	3465	(33.7)	(6.0)
Pennsylvania	6,708	(10.4)	2,879	1566	(15.5)	(1.7)
Rhode Island	146	(2.3)	67	34	(0.9)	(0.0)
South Carolina	5,158	36.7	1,542	1436	(19.8)	(4.6)
South Dakota	663	1.6	170	132	(0.3)	(0.1)
Tennessee	5,719	(27.6)	2,102	842	3.2	(2.1)
Texas	4,909	7.2	1,342	778	(7.3)	(5.3)
Utah	7,978	173.2	1,912	783	(20.3)	(0.1)
Vermont	1,822	(6.6)	850	495	6.7	(0.6)
Virginia	6,529	94.0	2,436	1360	(42.7)	(3.4)
Washington	8,951	15.6	6,035	2847	(16.9)	(6.0)
West Virginia	4,740	(28.5)	2,115	1034	(11.8)	(1.2)
Wisconsin	6,490	4.1	2,207	3363	(6.6)	(2.6)
Wyoming	4,632	29.7	1,701	501	(35.2)	(0.1)
Total	**251,558**	**903**	**93,145**	**56,697**	**(561)**	**(103)**

[1] soil carbon does not include effects of past land use history.

Net change reflects differences reported in the two most recent inventories per state.

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

	Age Class	SOC[1]	Dead Plant Matter	Biomass	***Total***
Region	*Years*		*Tg CO_2 eq.*		
North		**16,317**	**5,301**	**15,222**	**36,839**
	<20	1,473	224	295	**1,992**
	20-40	2,402	548	1,408	**4,357**
	40-60	4,757	1,487	4,286	**10,529**
	60-80	4,881	1,849	5,651	**12,380**
	80-100	2,042	866	2,631	**5,539**
	100-150	683	298	879	**1,859**
	150-200	36	14	32	**81**
	200+	4	1	5	**10**
	Uneven	40	15	35	**91**
South		**15,072**	**3,729**	**17,270**	**36,071**
	<20	4,944	703	2,451	**8,098**
	20-40	3,164	756	3,339	**7,259**
	40-60	3,064	958	4,747	**8,769**
	60-80	1,924	664	3,564	**6,153**
	80-100	574	196	1,094	**1,865**
	100-150	189	60	333	**581**
	150-200	7	2	10	**18**
	200+	1,205	390	1,731	**3,327**
Pacific Coast		**3,309**	**2,091**	**4,361**	**9,762**
	<20	777	247	196	**1,220**
	20-40	612	293	734	**1,639**
	40-60	627	414	1,023	**2,063**
	60-80	556	422	915	**1,893**
	80-100	370	314	654	**1,338**
	100-150	224	221	473	**919**
	150-200	51	62	145	**257**
	200+	38	47	97	**181**
	Uneven	54	71	125	**251**
Rocky Mountain		**1,554**	**1,587**	**1,897**	**5,038**
	<20	357	295	136	**789**
	20-40	114	89	71	**274**
	40-60	120	111	124	**355**
	60-80	226	224	295	**745**
	80-100	268	298	417	**983**
	100-150	310	374	572	**1,257**
	150-200	94	118	173	**384**
	200+	64	77	110	**251**
Total		**36,252**	**12,708**	**38,750**	**87,710**

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

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

	Age class	SOC[1]	Dead Plant Matter	Biomass	***Total***
Region	*Years*		*Tg CO_2 eq.*		
North		**7,548**	**1,942**	**5,271**	**14,761**
	<20	740	72	102	**915**
	20-40	983	153	372	**1,507**
	40-60	1,711	379	1,022	**3,111**
	60-80	2,252	673	1,918	**4,843**
	80-100	1,222	442	1,282	**2,946**
	100-150	570	199	510	**1,280**
	150-200	52	18	44	**114**
	200+	8	3	8	**19**
	Uneven	9	4	13	**26**
South		**2,659**	**718**	**3,485**	**6,862**
	<20	373	41	144	**558**
	20-40	407	77	321	**805**
	40-60	632	170	812	**1,614**
	60-80	710	237	1,189	**2,136**
	80-100	277	96	522	**895**
	100-150	110	41	232	**383**
	150-200	6	2	7	**15**
	Uneven	145	54	256	**455**
Pacific Coast		**4,881**	**4,357**	**10,060**	**19,298**
	<20	575	252	146	**973**
	20-40	499	240	495	**1,234**
	40-60	412	267	621	**1,300**
	60-80	663	529	1,241	**2,432**
	80-100	632	571	1,309	**2,512**
	100-150	860	892	2,105	**3,857**
	150-200	405	488	1,194	**2,087**
	200+	807	1,083	2,872	**4,761**
	Uneven	29	37	77	**142**
Rocky Mountain		**5,357**	**6,397**	**9,457**	**21,212**
	<20	907	739	310	**1,955**
	20-40	263	220	173	**655**
	40-60	213	201	239	**653**
	60-80	574	622	991	**2,188**
	80-100	861	1,018	1,680	**3,559**
	100-150	1,447	1,994	3,352	**6,792**
	150-200	729	1,072	1,810	**3,612**
	200+	364	531	903	**1,799**
Total		**20,445**	**13,414**	**28,273**	**62,132**

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

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

	Stand Size Class	SOC[1]	Dead Plant Matter	Biomass	***Total***
Region			*Tg CO* $_2$ *eq.*		
North		**22,630**	**6,844**	**19,478**	**48,952**
	Nonstocked	269	16	10	**295**
	Seedling/ Sapling	4,629	781	999	**6,408**
	Poletimber	7,797	2,158	5,337	**15,293**
	Sawtimber	9,935	3,889	13,132	**26,955**
South		**17,233**	**4,320**	**20,099**	**41,652**
	Nonstocked	197	7	16	**221**
	Seedling/ Sapling	4,412	623	1,708	**6,743**
	Poletimber	4,886	1,187	5,310	**11,383**
	Sawtimber	7,737	2,503	13,064	**23,305**
Pacific Coast		**6,546**	**4,819**	**11,123**	**22,489**
	Nonstocked	240	94	34	**368**
	Seedling/ Sapling	1,002	380	293	**1,675**
	Poletimber	756	377	777	**1,910**
	Sawtimber	4,548	3,968	10,019	**18,535**
Rocky Mountain		**3,661**	**4,450**	**7,186**	**15,297**
	Nonstocked	201	155	50	**406**
	Seedling/ Sapling	544	452	315	**1,312**
	Poletimber	711	729	1,180	**2,620**
	Sawtimber	2,205	3,114	5,642	**10,960**
Total		**50,070**	**20,433**	**57,886**	**128,390**

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

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

	Private	Public	Reserve/Other
Forest Type Group		*Tg CO_2 eq.*	
East	**41,194**	**9,547**	**2,196**
Aspen/Birch	1,020	644	95
Elm/Ash/Cottonwood	2,125	449	96
Loblolly/Shortleaf Pine	4,819	729	42
Longleaf/Slash Pine	821	262	13
Maple/Beech/Birch	7,705	1,965	658
Oak/Gum/Cypress	2,912	595	119
Oak/Hickory	15,787	2,918	730
Oak/Pine	3,162	643	103
Spruce/Fir	1,341	671	168
White/Red/Jack Pine	1,131	497	131
Other East Type Groups	372	174	41
West	**7,887**	**19,692**	**12,629**
Alder/Maple	451	195	15
Aspen/Birch	238	794	170
California Mixed Conifer	620	1,637	603
Douglas-fir	2,668	6,443	1,159
Fir/Spruce/Mt. Hemlock	607	4,712	2,498
Hemlock/Sitka Spruce	539	1,093	506
Lodgepole Pine	186	1,422	728
Other Western Hardwoods	120	78	293
Other Western Softwoods	30	271	305
Pinyon/Juniper	7	45	3,884
Ponderosa Pine	966	1,830	206
Redwood	208	13	108
Tanoak/Laurel	422	221	109
Western Larch	49	264	38
Western Oak	545	391	1,418
Western White Pine	1	23	45
Other West Type Groups	229	258	543
Total	**49,080**	**29,239**	**14,826**

[1] Excluding soils.

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

	Private	Public	Reserve/Other
Forest Type Group		$Tg\ CO_2\ eq.\ yr^{-1}$	
East	**(193.2)**	**(190.0)**	**(24.8)**
Aspen/Birch	15.7	0.5	1.9
Elm/Ash/Cottonwood	(16.0)	(17.5)	(1.1)
Loblolly/Shortleaf Pine	(58.5)	(10.8)	0.2
Longleaf/Slash Pine	(3.4)	(9.0)	(0.5)
Maple/Beech/Birch	(52.5)	(34.7)	13.0
Oak/Gum/Cypress	52.5	(20.3)	(7.5)
Oak/Hickory	(172.3)	(65.7)	(21.0)
Oak/Pine	26.9	(13.6)	(8.1)
Spruce/Fir	22.1	(1.5)	3.9
White/Red/Jack Pine	11.9	(12.8)	(7.7)
Other East Type Groups	(19.7)	(4.6)	2.2
West	**(56.6)**	**(190.3)**	**(96.4)**
Alder/Maple	-	-	-
Aspen/Birch	-	-	-
California Mixed Conifer	(62.1)	(124.0)	(33.4)
Douglas-fir	(7.5)	(46.0)	(0.9)
Fir/Spruce/Mt. Hemlock	(0.3)	(2.3)	14.3
Hemlock/Sitka Spruce	6.5	8.2	(17.8)
Lodgepole Pine	2.0	13.7	3.5
Other Western Hardwoods	-	-	-
Other Western Softwoods	(0.3)	(2.4)	(7.4)
Pinyon/Juniper	(0.8)	(2.3)	(51.6)
Ponderosa Pine	0.0	(26.0)	5.1
Redwood	5.3	2.2	(10.6)
Tanoak/Laurel	-	-	-
Western Larch	0.5	(11.0)	(1.3)
Western Oak	-	-	-
Western White Pine	(0.0)	(0.6)	3.6
Other West Type Groups	-	-	-
***Total*[2]**	**(249.8)**	**(380.3)**	**(121.2)**

[1] Excluding soils.

[2] Includes carbon storage in urban trees.

- Indicates no data available.