

## Chapter 3: Cropland Agriculture

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### 3.1 Sources of Greenhouse Gas Emissions in Cropland Agriculture

Cropland agriculture results in GHG emissions from multiple sources, with the magnitude of emissions determined, in part, by land management practices. Field burning of crop residues, cultivation of rice, and cultivation and management of soils leads to emissions of  $\text{N}_2\text{O}$ ,  $\text{CH}_4$ , and  $\text{CO}_2$ . However, agricultural soils can also mitigate GHG emissions through the biological uptake of organic carbon in soils resulting in  $\text{CO}_2$  removals from the atmosphere. This chapter covers both GHG emissions from cropland agriculture and biological uptake of  $\text{CO}_2$  in agricultural soils. National estimates of these sources, published in the U.S. GHG Inventory, are reported in this section and, where appropriate, State-level emissions estimates are provided.

#### 3.1.1 Residue Burning

Crop residues are sometimes burned in fields to prepare for cultivation and control for pests and disease, although this is not a common practice (EPA 2003a). While  $\text{CO}_2$  is a product of residue combustion, residue burning is not considered a net source of  $\text{CO}_2$  to the atmosphere. This is because  $\text{CO}_2$  released from burning crop biomass is replaced by uptake of  $\text{CO}_2$  in crops growing the following season (IPCC 1996). However,  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , also products of residue combustion, are not recycled back into crop biomass through biological uptake. Therefore, residue burning is considered a net source of  $\text{CH}_4$  and  $\text{N}_2\text{O}$  to the atmosphere. GHG emissions from field burning of crop residues are relatively small in the U.S. (EPA 2003a).

#### 3.1.2 Rice Cultivation

Rice cultivation is unique because it takes place almost universally on flooded fields and in the United States rice is grown exclusively on shallow, continuously flooded fields (EPA 2003a). This water regime causes  $\text{CH}_4$  emissions because waterlogged soils create conditions for anaerobic decomposition of organic matter, facilitated by  $\text{CH}_4$  emitting “methanogenic” bacteria (IPCC 1996).  $\text{CH}_4$  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  $\text{CH}_4$  emissions from rice cultivation, as are the availability of carbon substrate and other nutrients, soil pH, and partial pressure of  $\text{CH}_4$  (IPCC 1996). Because U.S. rice acreage is relatively small,  $\text{CH}_4$  emissions from rice cultivation are small relative to other cropland agriculture sources (EPA 2003a).

#### 3.1.3 Agricultural Soils

Agricultural soils, including cropland and grazing land, serve as both a source of GHGs and a mechanism to remove  $\text{CO}_2$  from the atmosphere. Both  $\text{N}_2\text{O}$  emissions and  $\text{CO}_2$  emissions and sinks are a function of underlying biological processes.  $\text{N}_2\text{O}$  is produced as an intermediate to natural nitrification and denitrification processes in the soil. In nitrification, soil microorganisms (“microbes”) convert ammonium to nitrate through aerobic oxidation (IPCC 1996). In denitrification, microbes convert nitrate to dinitrogen gas by anaerobic reduction. During

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nitrification and denitrification, soil microbes release N<sub>2</sub>O, which eventually reaches the atmosphere (IPCC 1996). Cropland soil amendments that add nitrogen to soils drive the production of N<sub>2</sub>O by providing additional substrate for nitrification and denitrification. Commercial fertilizer, livestock manure, sewage sludge, incorporation of crop residues, and cultivation of nitrogen-fixing crops all add nitrogen to soils. In addition, cultivating highly organic soils (i.e., histosols) enhances mineralization of nitrogen-rich organic matter, making more nitrogen available for nitrification and denitrification (EPA 2003a).

Nitrogen can be converted to N<sub>2</sub>O and emitted directly in agricultural fields, or it can be transported through groundwater and runoff to other systems where it is later converted to N<sub>2</sub>O, thus causing indirect emissions (IPCC 1996). Some applied nitrogen is volatilized into the atmosphere and subsequently deposited back onto soils, serving as additional, indirect sources of N<sub>2</sub>O (EPA 2003a). N<sub>2</sub>O from cropland soil amendments is the largest net source of U.S. GHG emissions from cropland agriculture (EPA 2003a).

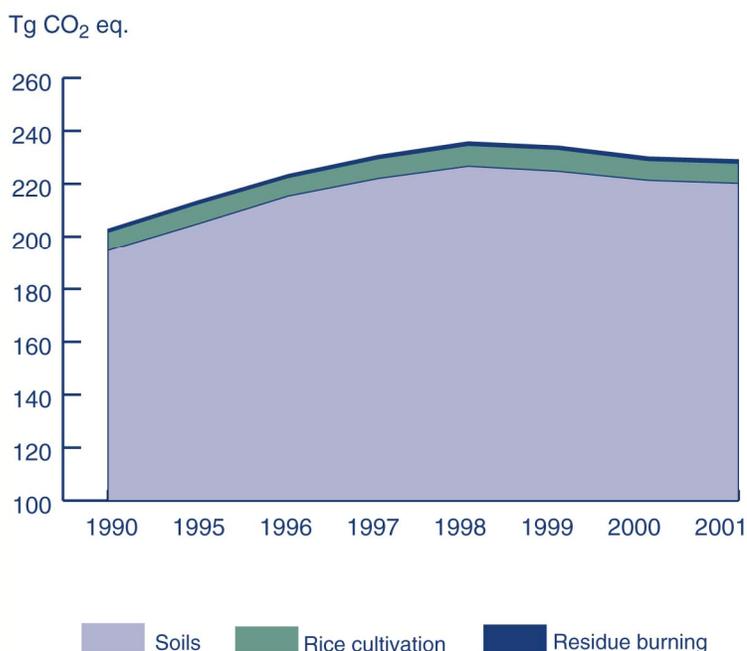
The size of CO<sub>2</sub> emissions and sinks in soils is related to the amount of organic carbon stored in soils (IPCC 1996). Changes in soil organic carbon content are related to inputs, e.g., atmospheric CO<sub>2</sub> fixed as carbon in plants through photosynthesis, and losses mainly driven by decomposition of soil organic matter causing CO<sub>2</sub> emissions (IPCC 1996). The net balance of CO<sub>2</sub> 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 affect the net balance of CO<sub>2</sub> through modifying inputs and rates of decomposition (IPCC 1996). Changes in agricultural practices such as clearing, drainage, tillage, crop selection, grazing, crop residue management, fertilization, and flooding can modify both organic matter inputs and decomposition, and thereby result in a net flux of CO<sub>2</sub> 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 3 percent in the upper 20-30 cm. 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 percent or more (Haas et al. 1957, Schlesinger 1986). However, 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 changes in agricultural management practices (Paustian et al. 1997ab, 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) will result in a net accumulation of soil organic carbon until a new equilibrium is achieved.

Cultivation of highly organic soils and additions of carbonate-based lime amendments cause

CO<sub>2</sub> losses to the atmosphere. Cultivated organic soils, also referred to as histosols, contain more than 20 to 30 percent organic matter by weight, and constitute a special case. Organic soils form under water-logged conditions, in which decomposition of plant residues is retarded. When organic soils are drained and cultivated the rate of decomposition and hence CO<sub>2</sub> emissions are greatly accelerated. Because of the depth and richness of the organic layers, carbon loss from cultivated organic soils can continue over long periods of time. Unless restored to undrained, anaerobic conditions, cultivated organic soils remain a net source of CO<sub>2</sub>.

Figure 3-1  
Trends in greenhouse gas emissions from cropland agriculture, 1990, 1995-2001



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<sub>2</sub> through the bicarbonate equilibrium reaction (IPCC 1996).

### 3.1.4 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 United States. In temperate systems, agroforestry practices store large amounts of carbon (Kort and Turlock 1999, Schroeder 1994), with the potential ranging from 15 to 198 tons of carbon per hectare (modal value of 34 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 is 90.3 million 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.

**Table 3-1 Summary of GHG emissions from cropland agriculture, 1990, 1995-2001**

Source	Gas	1990	1995	1996	1997	1998	1999	2000	2001
		<i>Tg CO<sub>2</sub> eq.</i>							
<i>Residue burning</i>	CH <sub>4</sub>	0.70	0.70	0.70	0.80	0.80	0.80	0.80	0.80
<i>Residue burning</i>	N <sub>2</sub> O	0.40	0.40	0.40	0.40	0.50	0.40	0.50	0.50
<i>Rice cultivation</i>	CH <sub>4</sub>	7.10	7.60	7.00	7.50	7.90	8.30	7.50	7.60
<i>Soils</i>	N <sub>2</sub> O	207.93	219.77	229.00	235.93	238.16	236.65	235.14	235.40
<i>Soils<sup>1</sup></i>	CO <sub>2</sub>	(13.30)	(14.90)	(13.60)	(13.90)	(11.50)	(11.90)	(13.80)	(15.20)
Mineral soils		(57.1)	(58.6)	(57.3)	(57.4)	(55.8)	(55.7)	(57.3)	(59.1)
Organic soils		34.3	34.8	34.8	34.8	34.8	34.8	34.8	34.8
Liming of soils		9.5	8.9	8.9	8.7	9.6	9.1	8.8	9.1
<i>Total emissions</i>		259.93	272.17	280.80	288.13	291.76	290.05	287.54	288.20
<i>Net emissions (sources and sinks)</i>		202.83	213.57	223.50	230.73	235.86	234.25	230.14	229.10

Note: Parentheses indicate net sequestration.

<sup>1</sup> Soil carbon sequestration on land under the Conservation Reserve Program and on range and grazing lands is included in the total for mineral soils.

### 3.2 Summary of U.S. Greenhouse Gas Emissions from Cropland Agriculture

In 2001, cropland agriculture resulted in total emissions of 288 Tg CO<sub>2</sub> eq. of GHG (Table 3-1). GHG emission from agricultural soils, including N<sub>2</sub>O and CO<sub>2</sub>, were responsible for the majority of total emissions, while residue burning and rice cultivation caused less than 4 percent of emissions (Table 3-1). N<sub>2</sub>O and CO<sub>2</sub> from agricultural soils totaled 235 Tg CO<sub>2</sub> eq. and 44 Tg CO<sub>2</sub> eq., respectively, in 2001. However, that amount was offset by the uptake of 59 Tg CO<sub>2</sub> eq. in agricultural soils in 2001. Thus, net emissions of GHGs from cropland agriculture were just under 230 Tg CO<sub>2</sub> eq. Since 1990, the magnitude of residue burning and rice cultivation emissions has remained relatively stable (Figure 3-1). Comparatively, GHG emissions from agricultural soils increased from 1990 to 1998 then decreased each year until 2001 (Table 3-1). Overall, net GHG emissions from cropland agriculture increased 12 percent from 1990 and 2001.

### 3.3 Residue Burning

GHG emissions from field burning of crop residues are a function of the amount and type of residues burned. In the United States, crops burned include wheat, rice, sugarcane, corn, barley, soybeans, and peanuts (EPA 2003a). For most crops, less than 5 percent of residues are burned

**Table 3-2 Greenhouse gas emissions from agriculture burning, by crop type, 1990-2001**

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
	<i>Tg CO<sub>2</sub> eq.</i>											
<b>CH<sub>4</sub></b>	0.68	0.64	0.75	0.60	0.81	0.66	0.75	0.76	0.78	0.76	0.78	0.76
Wheat	0.14	0.10	0.12	0.12	0.12	0.11	0.11	0.12	0.13	0.12	0.11	0.10
Rice	0.08	0.08	0.08	0.08	0.10	0.08	0.09	0.07	0.06	0.07	0.07	0.07
Sugarcane	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Corn	0.28	0.27	0.34	0.23	0.36	0.26	0.33	0.33	0.35	0.34	0.35	0.34
Barley	0.02	0.02	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Soybeans	0.15	0.15	0.17	0.14	0.19	0.17	0.18	0.21	0.21	0.20	0.21	0.22
Peanuts	-	-	-	-	-	-	-	-	-	-	-	-
<b>N<sub>2</sub>O</b>	0.37	0.36	0.41	0.34	0.45	0.38	0.42	0.45	0.45	0.44	0.46	0.46
Wheat	0.05	0.03	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.03
Rice	0.04	0.04	0.04	0.04	0.05	0.04	0.04	0.03	0.03	0.03	0.03	0.03
Sugarcane	-	-	-	-	-	-	-	-	0.01	0.01	0.01	0.01
Corn	0.09	0.08	0.11	0.07	0.11	0.08	0.10	0.10	0.11	0.11	0.11	0.11
Barley	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	-	-	-
Soybeans	0.18	0.19	0.21	0.18	0.24	0.21	0.23	0.26	0.26	0.25	0.26	0.28
Peanuts	-	-	-	-	-	-	-	-	-	-	-	-
<b>Total</b>	1.05	1.00	1.16	0.94	1.26	1.03	1.17	1.21	1.24	1.20	1.24	1.22

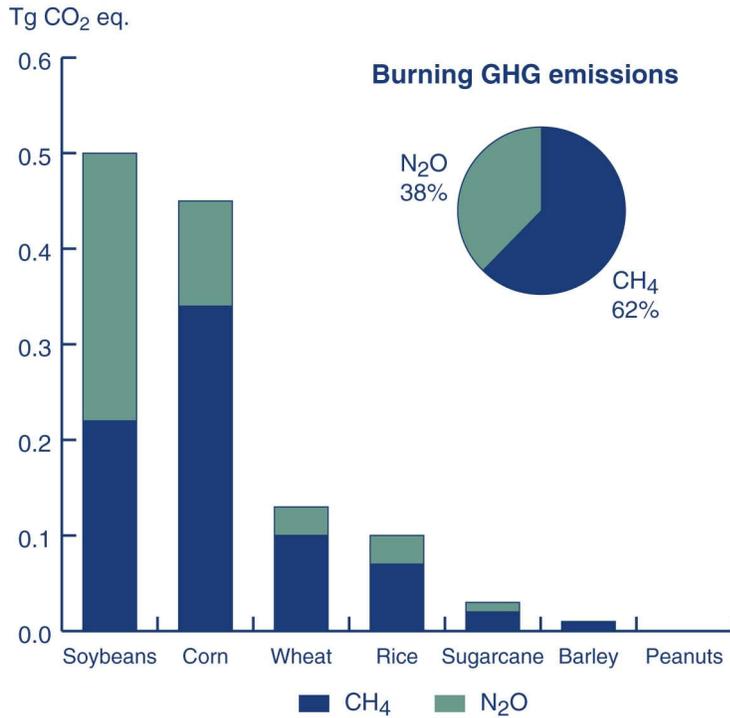
- = less than 0.005

per year; a higher portion of rice residues is burned annually (EPA 2003a).

About two-thirds of GHG emissions from residue burning were CH<sub>4</sub> across all crop types, in 2001; the remaining third were N<sub>2</sub>O (Table 3-2; Figure 3-2). The highest GHG emissions were from burning of soybean and corn crop residues, at 40 percent and 30 percent respectively. Burning of wheat, rice, sugarcane, and barely crop residues each contributed 10 percent or less to overall GHG emissions; burning of peanut crop residues contributed almost nothing to this source of GHGs.

Total GHG emissions from residue burning increased 17 percent from 1990 to 2001. Trends in relative GHG emissions were similar across crop types in 1990 compared to 2001 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 (Figure 3-3). By

Figure 3-2  
**Greenhouse gas emissions from burning by crop type, 2001**



2001, these positions reversed, corresponding to changes in production. Between 1990 and 2001, soybean and corn production both increased in absolute amounts [Figure 3-4(A)]. However, proportionally, soybean production increased more dramatically than corn (soybean production increased by 50 percent and corn by 20 percent) [Figure 3-4(B)]. In addition, soybeans have higher nitrogen content than corn, resulting in greater N<sub>2</sub>O emission per unit of crop mass burned (Appendix Table B-1). Thus, while corn production was still greater than soybean production in 2001, GHG emissions from soybean residue burning exceeded those from corn residue burning. Appendix Table B-2 provides the complete time series of crop production from 1990 to 2001 for crop types that contribute to GHG emissions from burning.

Illinois and Iowa had the highest State levels of GHG emissions from residue burning in 2001, with each emitting roughly 0.16 Tg CO<sub>2</sub> eq. of CH<sub>4</sub> and N<sub>2</sub>O combined (Appendix Table B-4 and Appendix Table B-5). The next highest levels of GHG emissions from residue burning were in Nebraska, Indiana, Minnesota, Arkansas, Ohio, Kansas, Missouri, and South Dakota, with emissions between 0.04 and 0.1 Tg CO<sub>2</sub> 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.

### 3.4 Methods for Estimating CH<sub>4</sub> and N<sub>2</sub>O Emissions from Residue Burning

EPA provided national-level estimates of GHG emissions from agricultural residue burning for all crop types except rice, 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 5 of the U.S. GHG Inventory report (EPA 2003a). This

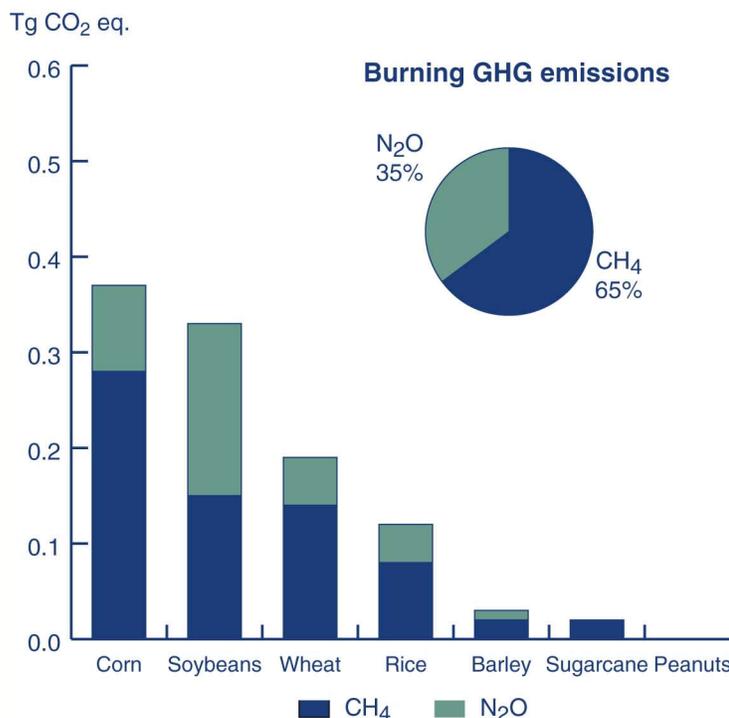
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The equations below were used to estimate the amounts of carbon and nitrogen released during burning. Final emissions estimates were derived from the amount of carbon and nitrogen released using emissions ratios for CH<sub>4</sub>, CO, N<sub>2</sub>O, and NO<sub>x</sub> published by the IPCC.

$$\begin{aligned} \text{Carbon Released} = & (\text{Annual Crop Production}) \times (\text{Residue/Crop Product Ratio}) \\ & \times (\text{Fraction of Residues Burned in situ}) \times (\text{Dry Matter Content of the Residue}) \\ & \times (\text{Burning Efficiency}) \times (\text{Carbon Content of the Residue}) \times (\text{Combustion Efficiency}) \end{aligned}$$

$$\begin{aligned} \text{Nitrogen Released} = & (\text{Annual Crop Production}) \times (\text{Residue/Crop Product Ratio}) \\ & \times (\text{Fraction of Residues Burned in situ}) \times (\text{Dry Matter Content of the Residue}) \\ & \times (\text{Burning Efficiency}) \times (\text{Nitrogen Content of the Residue}) \times (\text{Combustion Efficiency}) \end{aligned}$$

Figure 3-3  
**Greenhouse gas emissions from burning by crop type, 1990**



Values used in the above equations to estimate emissions from residue burning are summarized in Appendix Table B-1. 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:* 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 1992b). Crop production data for these crops, except rice in Florida, were taken from USDA NASS (Field Crops: 1994, 1998; Crop Production 2001, 2000). Rice production data for Florida were estimated by applying average primary and ratoon crop yields for Florida (Smith 1999) to Florida acreages (Schueneman 1999b, 2001; Deren 2002).

Figure 3-4(A)  
**Change in commodity production, 1990-2001**

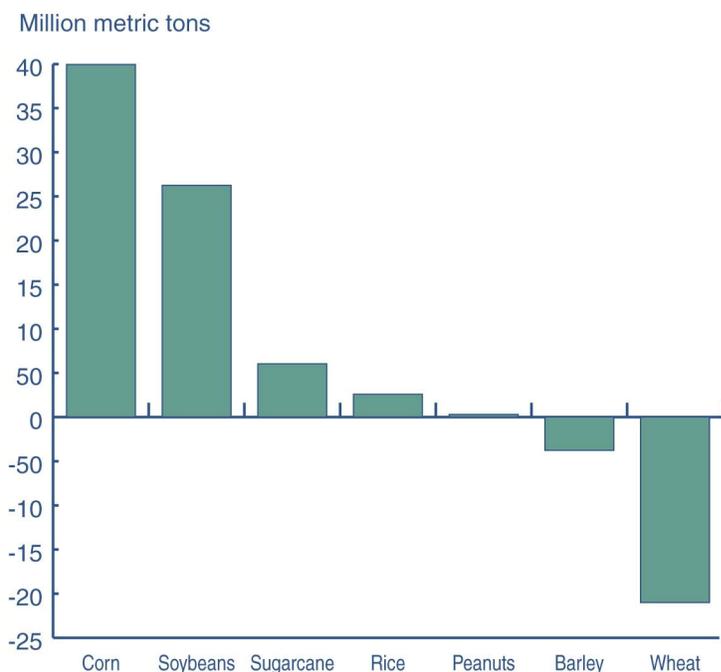
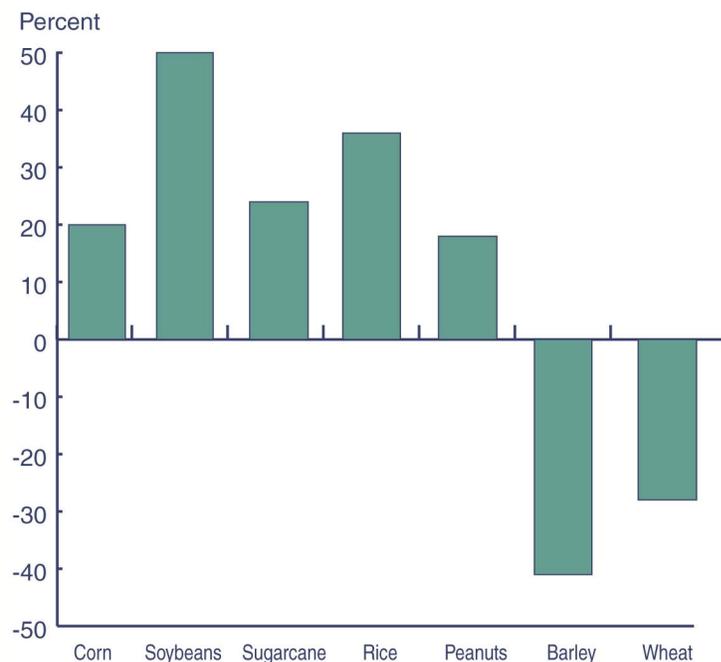


Figure 3-4(B)  
**Percent change in commodity production, 1990-2001**



*Residue-to-Crop Product Mass Ratios:* All residue/crop product mass ratios except sugarcane were obtained from Strehler and Stützel (1987). 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 3 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, and 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 1999a, 1999b, 2000, 2001, 2002; Wilson 2001, 2002) (Appendix Table B-1).

The estimates provided for Arkansas and Florida remained constant over the entire 1990-2001 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

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burned in Sacramento Valley is representative of burning in the entire State, because the Sacramento Valley accounts for over 95 percent 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 2001 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 Stütze (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 percent.

*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<sub>2</sub>. Combustion efficiency was assumed to be 88 percent for all crop types (EPA 1994).

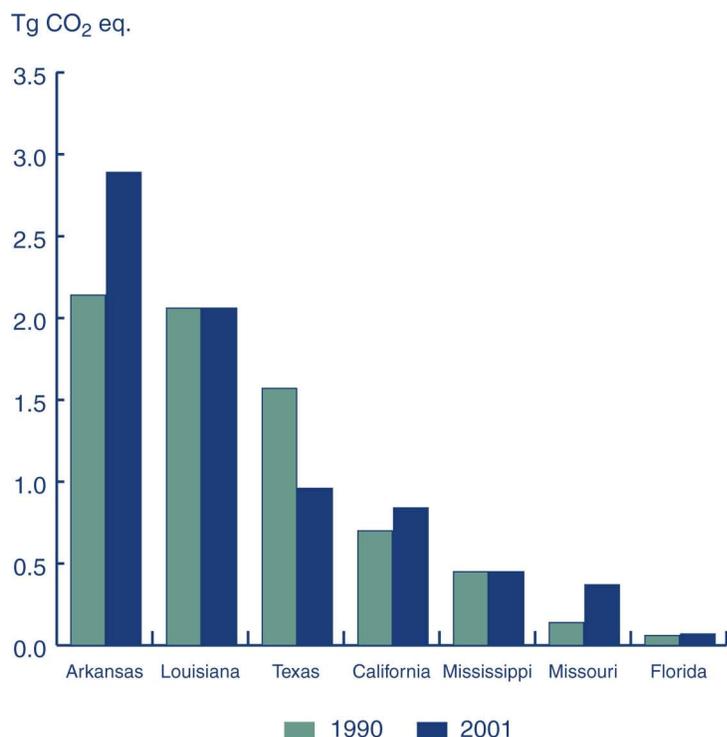
State-level emissions estimates were calculated with the above equations, applying State-level production data to national-level coefficients. The State-level rice estimates were provided directly by EPA, using State-specific residue fractions for rice because the fraction of residues burned varies among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

### 3.5 Uncertainty in Estimating CH<sub>4</sub> and N<sub>2</sub>O 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. 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, are 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 were based upon information collected by State agencies and in the published literature. These emissions

Figure 3-5  
**Methane from rice cultivation by State,  
 1990 and 2001**



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.

### 3.6 Rice Cultivation

CH<sub>4</sub> emissions from rice cultivation<sup>10</sup> are limited to seven U.S. States. In four States, the climate allows for cultivation of two rice crops per season, the second of which is referred to as a ratoon crop (EPA 2003a). CH<sub>4</sub> emissions from primary and ratoon crops are accounted for separately because emissions are higher from ratoon crops (EPA 2003a). Overall, rice cultivation is a small source of CH<sub>4</sub> in the United States. In 2001, CH<sub>4</sub> emissions totaled 7.6 Tg CO<sub>2</sub> eq, of which 5.9 Tg CO<sub>2</sub> eq. were from primary crops in all seven States and 1.7 Tg CO<sub>2</sub> were from ratoon crops in four States (Appendix Table B-7).

Arkansas and Louisiana had the highest CH<sub>4</sub> emissions from rice cultivation in 2001, followed by Texas and California. Mississippi, Missouri, and Florida each had emissions of less than 0.5 Tg CO<sub>2</sub> eq. (Figure 3-5). Since 1990, CH<sub>4</sub> emissions from rice cultivation have increased 7 percent. While small national changes were seen between 1990 and 2001, sizeable shifts occurred at State levels during that time period. For example, CH<sub>4</sub> emission in Arkansas and California increased by 35 percent and 19 percent, respectively, while emissions in Texas declined by 39 percent (Figure 3-5 and Table 3-3). CH<sub>4</sub> emissions from Missouri increased by

<sup>10</sup> This source focuses on CH<sub>4</sub> emissions resulting from anaerobic decomposition, and does not include emissions from burning of rice residues. The later is covered in section 3.3.

over 150 percent between 1990 and 2001, but remained small in magnitude relative to emissions from other States. State-level shifts in CH<sub>4</sub> emissions since 1990 are positively correlated with changes in area of rice cultivation (Appendix Table B-6). Appendix Table B-6 provides a complete time series of areas harvested for rice by State and primary versus ratoon crops from 1990-2001.

### 3.7 Methods for Estimating CH<sub>4</sub> Emissions from Rice Cultivation

EPA provided estimates for CH<sub>4</sub> emissions from rice cultivation for this report. Details on the methods are provided below and are excerpted, with permission from EPA, from Chapter 5 of the U.S. GHG Inventory report (EPA 2003a). The method used by EPA applies area-based seasonally integrated emission factors (i.e., amount of CH<sub>4</sub> emitted over a growing season per unit harvested area) to harvested rice areas to estimate annual CH<sub>4</sub> emissions from rice cultivation. EPA derived specific CH<sub>4</sub> emission factors from published studies containing rice field measurements in the United States, with separate emissions factors for ratoon and primary crops to account for higher seasonal emissions in ratoon crops.

A review of published experiments was used to develop emissions factors for primary and ratoon crops. Experiments where nitrate or sulfate fertilizers or other substances believed to suppress CH<sub>4</sub> 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<sub>4</sub>/hectare-season, and the resultant emission factor for the ratoon crop is 780 kg CH<sub>4</sub>/hectare-season.

The harvested rice areas for the primary and ratoon crops in each State are presented in Appendix Table B-6. Primary crop areas for 1990 through 2001 for all States except Florida

**Table 3-3 Change in methane emissions from rice cultivation, 1990-2001**

	1990	2001	Change, 1990-2001
	<i>Tg CO<sub>2</sub> eq.</i>		<i>percent</i>
Arkansas	2.14	2.89	35
California	0.70	0.84	19
Florida	0.06	0.07	4
Louisiana	2.06	2.06	0
Mississippi	0.45	0.45	1
Missouri	0.14	0.37	159
Texas	1.57	0.96	-39
Total	7.12	7.64	7

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were taken from USDA NASS Field Crops Final Estimates 1987-1992 (USDA NASS Field Crops 1994), Field Crops Final Estimates 1992-1997 (USDA NASS Field Crops 1998), Crop Production 2000 Summary (USDA NASS Crop Production 2001), and Crop Production 2001 Summary (USDA NASS Crop Production 2002). Harvested rice areas in Florida, which are not reported by USDA, were obtained from Tom Schueneman (1999b, 1999c, 2000, 2001a), 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 percent of the primary area (Slaton 1999, 2000, 2001a). In Florida, the ratoon area was 50 percent of the primary area from 1990 to 1998 (Schueneman 1999a), about 65 percent of the primary area in 1999 (Schueneman 2000), around 41 percent of the primary area in 2000 (Schueneman 2001a), and about 70 percent of the primary area in 2001 (Deren 2002). In Louisiana, the percentage of the primary area in ratoon was constant at 30 percent over the 1990 to 1999 period, but increased to approximately 40 percent in 2000, before returning to 30 percent in 2001 (Linscombe 1999a, 2001a, 2002 and Bollich 2000). In Texas, the percentage of the primary area in ratoon was constant at 40 percent over the entire 1990 to 1999 period and in 2001, but increased to 50 percent in 2000 due to an early primary crop (Klosterboer 1999, 2000, 2001a, 2002).

### 3.8 Uncertainty in Estimating CH<sub>4</sub> 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. The information is reproduced here with permissions from EPA.

CH<sub>4</sub> 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, and 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<sub>4</sub>/hectare-season and ratoon emissions ranged from 481 to 1,490 kg CH<sub>4</sub>/hectare-season. Based on these emission ranges, total CH<sub>4</sub> emissions from rice cultivation in 2001 were estimated to range from 1.7 to 17 Tg CO<sub>2</sub> eq.

In addition, data are not collected regularly on the area of rice crops in ratoon, creating another relatively minor source of uncertainty. The area estimates are derived from expert opinion and account for less than 10 percent 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 5 to 68 percent 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.

### 3.9 Agricultural Soils

Sources of N<sub>2</sub>O are addressed in this section, including soil amendments (commercial fertilizer, livestock manure, crop residues, and sewage sludge), nitrogen-fixing crops, and histosol cultivation. National-level emissions estimates are discussed; State-level data for these sources are not yet available, although work is underway to develop process models that can make finer-scale estimates of N<sub>2</sub>O and other GHG emissions from agriculture (Box 3-1). CO<sub>2</sub> emissions and sinks in agricultural soils are discussed in the following section.

#### 3.9.1 Commercial Fertilizer

Commercial fertilizers were the largest source of N<sub>2</sub>O emissions in cropland soils in 2001, emitting over 100 Tg CO<sub>2</sub> eq. of N<sub>2</sub>O (Figure 3-6, Table 3-4). The majority of emissions from commercial fertilizers were direct, while some were from leaching, and a small portion was from volatilization. The methods used to estimate emissions from fertilizer assume that all fertilizer purchased in the United States is used in cropland agriculture, which is a simplifying assumption necessitated by limited data. Thus, the numbers reported here and in the U.S. GHG Inventory could overestimate emissions attributable to cropland soils. Total N<sub>2</sub>O emissions from fertilizer are more likely apportioned across a range of land uses where fertilizer is applied. For example, fertilizer is applied to forests and lawns—some estimates suggest as much as 6 percent of U.S. fertilizer consumption is for turfgrass (EPA 1999).

#### Box 3-1

### National Inventory Development and Assessment of Soil-Atmosphere Exchange of N<sub>2</sub>O, NO<sub>x</sub> and CH<sub>4</sub> Oxidation in U.S. Agricultural Soils

A.R. Mosier (USDA – ARS), W.J. Parton (Colo. St. Univ.), S. J. Del Grosso (Colo. St. Univ.) and T. Wirth (EPA)

A collaborative agreement between EPA and USDA/ARS and the Colorado State University Natural Resource Ecology Laboratory is laying the groundwork for using the ecosystem-process-based model DAYCENT to conduct the U.S. national inventory for N<sub>2</sub>O emissions from agricultural soils. Over the next few years the model-based inventory will be developed and compared and contrasted to the IPCC Agricultural Soils N<sub>2</sub>O estimation that is now used in the national inventory.

#### DAYCENT Model Description

The DAYCENT ecosystem model (Parton et al., 1998; Kelly et al., 2000; Del Grosso et al., 2001) simulates exchanges of carbon (C), nutrients (N, P, S), and gases (CO<sub>2</sub>, CH<sub>4</sub>, N<sub>2</sub>O, NO<sub>x</sub>, N<sub>2</sub>) among the atmosphere, soil, and plants. Required inputs to drive the model include daily maximum/minimum temperature and precipitation, site-specific soil properties, and current and historical land use. Disturbances and management practices such as fire, grazing, cultivation, and organic matter or fertilizer additions can be simulated. The submodels used in DAYCENT are described in detail by Del Grosso et al. (2001) and the model code is available from the authors.

DAYCENT includes submodels for plant productivity, decomposition of dead plant material and soil organic matter (SOM), soil water and temperature dynamics, and trace gas fluxes. Flows of C and nutrients are controlled by the amount of C in the various pools (e.g. SOM, plant biomass), the N and lignin concentrations of the pools, abiotic temperature/soil water factors, and soil physical properties related to texture. SOM is divided into three pools based on decomposition rates (Parton et al., 1993; 1994). Decomposition of SOM and external nutrient additions supply the nutrient pool available for plant growth and microbial processes resulting in trace gas fluxes. Plant growth is controlled by: a plant-specific

*(Continued on page 58)*

(Continued from page 57)

maximum growth parameter, nutrient availability, and 0-1 multipliers that reflect shading, water, and temperature stress. Net Primary Productivity (NPP) is allocated among leafy, woody, and root compartments as a function of plant type, season, soil water content, and nutrient availability (Metherell et al., 1993). The land surface submodel of DAYCENT simulates water flow and evapotranspiration for the plant canopy, litter, and soil profile, as well as soil temperature throughout the profile (Parton et al., 1998; Eitzinger et al., 2000).

The trace gas submodel of DAYCENT simulates N<sub>2</sub>O, NO<sub>x</sub>, and N<sub>2</sub> emissions from soils resulting from nitrification and denitrification as well as CH<sub>4</sub> oxidation in soils. The nitrification submodel simulates N<sub>2</sub>O and NO<sub>x</sub> emissions as a function of soil NH<sub>4+</sub>, water content, temperature, pH, and texture (Parton et al., 2001). Nitrification is limited by moisture stress when soil water-filled pore space (WFPS) is too low and by O<sub>2</sub> availability when WFPS is too high. Optimum WFPS for nitrification is ~55%, with a higher optimum for clay than sandy soils.

The denitrification submodel simulates N<sub>2</sub>O, N<sub>2</sub>, and NO<sub>x</sub> emissions as a function of soil NO<sub>3-</sub>, water content, labile C availability (most denitrifiers are heterotrophs), and soil physical properties related to texture that influence gas diffusion rates (Del Grosso et al., 2000b). Denitrification, an anaerobic process, does not occur until WFPS exceeds 50-60 percent then it increases exponentially as WFPS increases and levels off as soils approach saturation. Simulated heterotrophic respiration rates are used as a proxy for labile C availability.

NO<sub>x</sub> emissions are calculated using total N<sub>2</sub>O emissions, a NO<sub>x</sub>:N<sub>2</sub>O function based on soil gas diffusivity, and a pulse multiplier based on rainfall frequency and amount (Parton et al., 2001). As soil gas diffusivity decreases, a smaller proportion of total N gas fluxes are assumed to be in the form of NO<sub>x</sub> because NO<sub>x</sub> becomes more reactive as soils become more reducing. The pulse multiplier equations were developed by Yienger and Levy (1995) and account for the observed high NO<sub>x</sub> emission rates following precipitation events onto soils that were previously dry (Smart et al., 1999; Martin et al., 1998; Hutchinson et al., 1993). CH<sub>4</sub> uptake is controlled by soil gas diffusivity, water content, and temperature (Del Grosso et al., 2000a). CH<sub>4</sub> oxidation rates are assumed to be limited by gas diffusivity when volumetric soil water content is too high and by moisture stress on biological activity when volumetric soil water content is too low. Optimum volumetric soil water content values range from 0.06-0.22 cm<sup>3</sup> H<sub>2</sub>O per cm<sup>3</sup> soil. As with nitrification, clay soils are assumed to have higher optimum water content for CH<sub>4</sub> oxidation than sandy soils.

In 2001, total commercial synthetic fertilizer consumption was 10,684 Gg<sup>11</sup> of nitrogen, an increase of 580 Gg of nitrogen (or 6 percent) since 1990. The highest levels of commercial synthetic fertilizer consumption, and therefore emissions, were in 1999 at 11,237 Gg of nitrogen [Figure 3-7 (A)]. Since then, consumption has decreased steadily. Even so, nitrogen applications of commercial synthetic fertilizers were still far greater than those from other sources in cropland soils in 2001.

### 3.9.2 Livestock Manure

Livestock manure application was the fourth largest source of N<sub>2</sub>O emissions from cropland soils in 2001 (Figure 3-6, Table 3-4). About half of the 28 Tg CO<sub>2</sub> eq. of N<sub>2</sub>O from livestock waste was emitted directly (14 Tg CO<sub>2</sub> eq.). The remaining portion was predominantly indirect

<sup>11</sup> By convention, activity data are expressed in Gg given their magnitude. One Gg equals 10<sup>-3</sup> Tg. Box 1-1 gives additional information on units.

from leaching and runoff (11 Tg CO<sub>2</sub> eq.) and a small portion was from volatilization (3 Tg CO<sub>2</sub> eq.). N<sub>2</sub>O from livestock waste applied to croplands increased 11 percent between 1990 and 2001, with steady increases most years (Table 3-4) tracking the trend in livestock waste application to crops [Figure 3-7 (B)]. While livestock manure is a source of N<sub>2</sub>O when applied to soils, it also enhances soil carbon sequestration by providing an input of organic matter to soils. Carbon gains from livestock waste applications essentially offset a portion of GHG emissions. The size of this offset is addressed in the next section on CO<sub>2</sub> emissions and sinks in agricultural soils.

### 3.9.3 Nitrogen Fixation

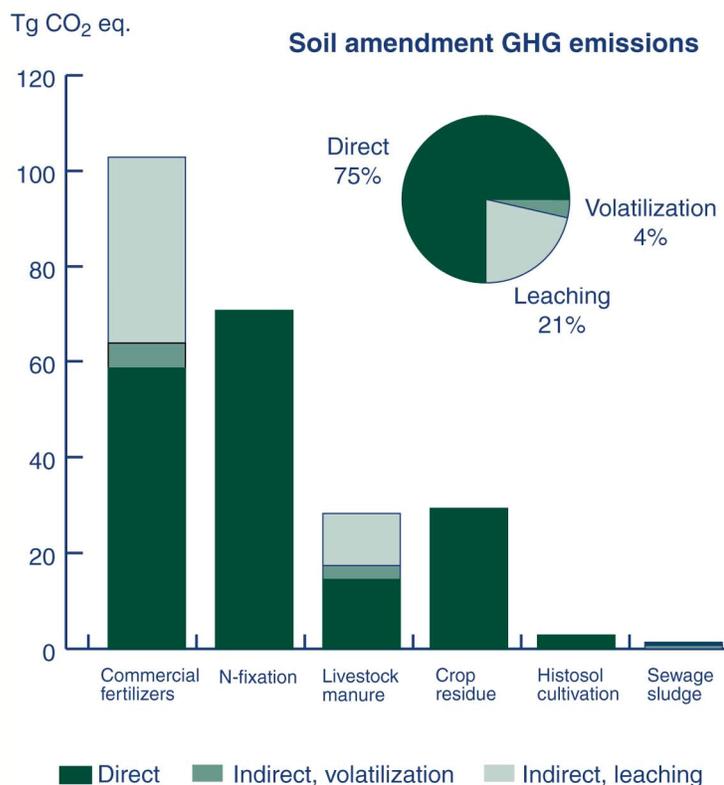
Nitrogen-fixing plants were the second largest source of N<sub>2</sub>O emissions in cropland soils, contributing about 70 Tg CO<sub>2</sub> eq. in 2001 (Figure 3-6). Nitrogen

additions, which are related to nitrogen content in aboveground plant biomass, vary by crop type and production levels (EPA 2003a). While numerous nitrogen fixers are cultivated, emissions from this source are largely attributable to three species of plant: soybeans, white clover, and alfalfa. In 2001, total aboveground biomass nitrogen was highest in soybeans, second highest in white clover, and third highest in alfalfa. Combined, these three crops contributed 93 percent (10,745 Gg) of all nitrogen from nitrogen-fixing crops to soils (Appendix Table B-9). Soybeans and alfalfa are grown in croplands, while white clover is typically grown on pastures for forage. Additional nitrogen fixers include peanuts, dry edible beans, edible peas, lentils, wrinkled seed peas, and Australian winter peas. Red clover, birdsfoot trefoil, arrowleaf clover, and crimson clover are nitrogen fixers grown as forage on pastures.

### 3.9.4 Crop Residues

The incorporation of crop residues into soils led to N<sub>2</sub>O emissions of 29 Tg CO<sub>2</sub> eq. in 2001.

Figure 3-6  
**Nitrous oxide emissions from agricultural soil management by source and process, 2001**



**Table 3-4 Nitrous oxide emissions from agricultural soil amendments, 1990-2001**

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
<i>Tg CO<sub>2</sub> eq.</i>												
<b><i>Commercial fertilizers</i></b>	97.26	98.93	99.66	103.18	107.45	103.99	107.46	107.63	107.81	108.22	104.94	102.88
Direct emissions	55.40	56.35	56.76	58.77	61.20	59.23	61.20	61.30	61.40	61.64	59.77	58.60
Volatilization emissions	4.93	5.01	5.05	5.23	5.44	5.27	5.45	5.46	5.47	5.49	5.32	5.21
Leaching and run-off emissions	36.93	37.57	37.84	39.18	40.81	39.49	40.81	40.87	40.94	41.10	39.85	39.07
<b><i>Livestock manure</i></b>	25.44	26.12	26.01	26.55	26.74	26.46	26.76	27.35	27.68	27.65	28.14	28.27
Direct emissions	13.03	13.38	13.32	13.60	13.69	13.55	13.71	14.01	14.17	14.16	14.41	14.48
Volatilization emissions	2.64	2.71	2.70	2.75	2.77	2.74	2.78	2.84	2.87	2.87	2.92	2.93
Leaching and run-off emissions	9.77	10.03	9.99	10.20	10.27	10.17	10.28	10.50	10.63	10.62	10.81	10.86
<b><i>Sewage sludge</i></b>	0.74	0.83	0.93	1.04	1.13	1.23	1.26	1.28	1.30	1.35	1.40	1.40
Direct emissions	0.38	0.43	0.48	0.53	0.58	0.63	0.65	0.66	0.67	0.69	0.72	0.72
Volatilization emissions	0.08	0.09	0.10	0.11	0.12	0.13	0.13	0.13	0.13	0.14	0.14	0.14
Leaching and run-off emissions	0.28	0.32	0.36	0.40	0.44	0.47	0.48	0.49	0.50	0.52	0.54	0.54
<b><i>N fixation (direct emissions)</i></b>	58.46	59.52	61.39	57.08	66.07	61.84	63.86	68.15	69.22	68.25	68.80	70.62
<b><i>Crop residue (direct emissions)</i></b>	23.22	22.46	26.34	21.11	28.11	23.41	26.80	28.67	29.29	28.31	28.97	29.33
<b><i>Histosol cultivation (direct emissions)</i></b>	2.81	2.81	2.80	2.82	2.83	2.84	2.85	2.86	2.87	2.88	2.89	2.90

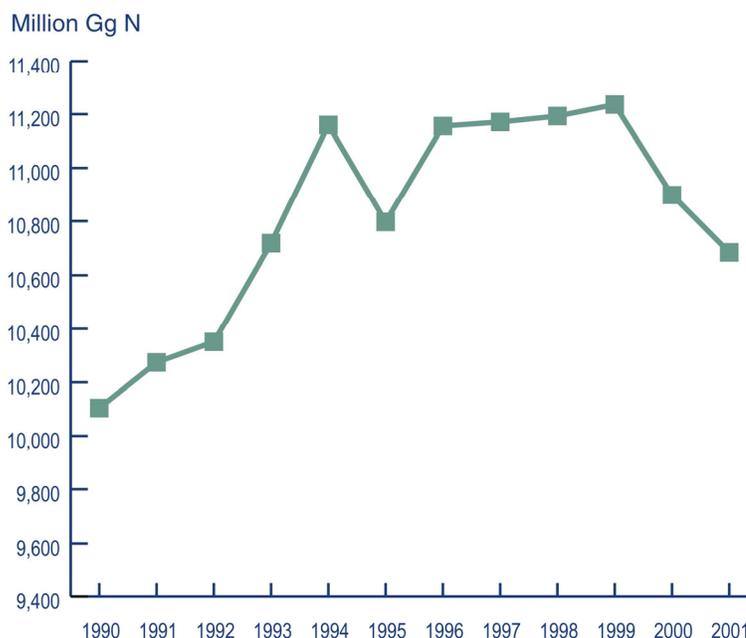
Soybean residues were by far the largest source of nitrogen, adding 2,975 Gg N to soils in 2001 (Appendix Table B-10). Corn was the next largest source at 1,147 Gg N, or about a third that of soybeans. Wheat, sorghum, rice, barley, peanuts, and oats each added between 10 and 400 Gg N, while dry edible peas, rye, lentils, wrinkled seed peas, and Australian winter peas each contributed less than 10 Gg N.

Soybeans contribute to N<sub>2</sub>O emissions both from nitrogen fixation and incorporation of crop residues in the soil. The total contribution of soybeans to GHG emissions increased between 1990 and 2001 as evidenced by the 50 percent increase in nitrogen inputs in both aboveground biomass and crop residues from soybeans (Appendix Table B-9 and Appendix Table B-10). The increase is directly related to increased production of soybeans nationwide. In 2001 soybean production was over 78 million metric tons, up from 52 million metric tons in 1990 (Appendix Table B-2).

### 3.9.5 Histosol Cultivation

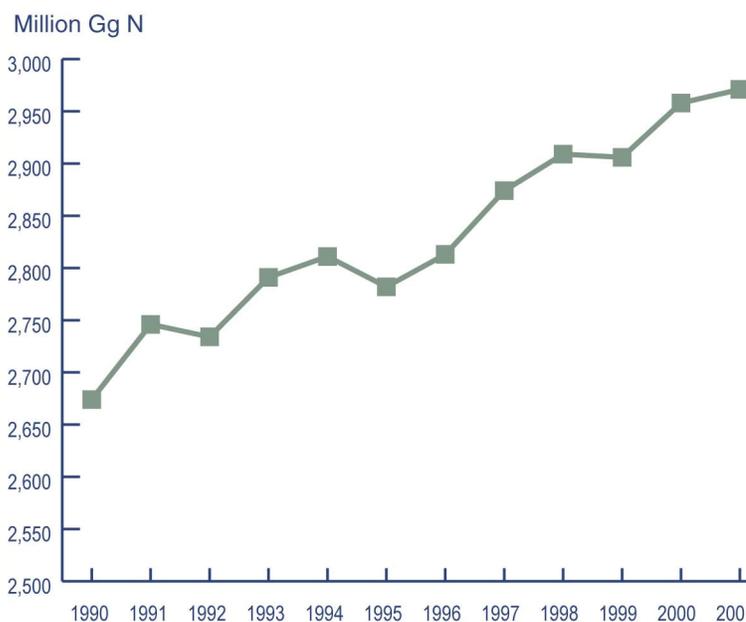
Histosol cultivation is a relatively minor source of N<sub>2</sub>O emissions from cropland soils. In 2001, N<sub>2</sub>O emissions from this source were 2.9 Tg CO<sub>2</sub> eq, up from 2.8 Tg CO<sub>2</sub> eq. in 1990 (Table 3-4). Sewage sludge is another minor source of N<sub>2</sub>O emissions from cropland soils, with

Figure 3-7(A)  
**Commercial synthetic fertilizer consumption, 1990-2001**



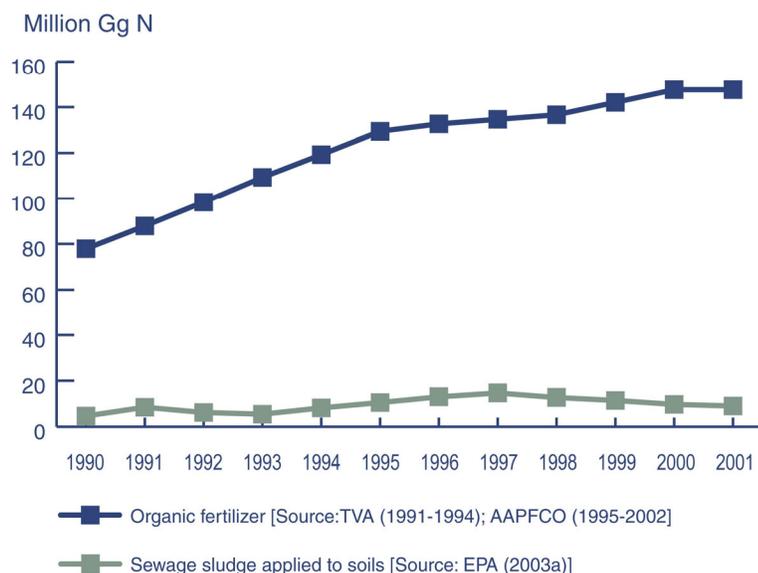
Source: TVA (1991-1994); AAPFCO (1996-2002)

Figure 3-7(B)  
**Livestock manure applied, 1990-2001**



Source: EPA (2003a)

Figure 3-7(C)  
**Commercial organic fertilizer and sewage sludge applied, 1990-2001**



direct and indirect emissions totaling 1.4 Tg CO<sub>2</sub> eq. in 2001. While small in overall magnitude, 2001 levels of N<sub>2</sub>O from sewage sludge were 90 percent higher than 1990 levels (Table 3-4). Over this time period, more sludge was applied to land rather than disposing of it in landfills, dumping it into oceans, or burning (EPA 2003a).

### 3.10 Methods for Estimating N<sub>2</sub>O Emissions from Agricultural Soils

Emissions of N<sub>2</sub>O 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 2003a), with one

exception. The U.S. GHG Inventory includes in Agricultural Soils Management direct and indirect emissions of N<sub>2</sub>O from livestock manure deposited on pasture, range, and paddock, while the USDA GHG Inventory includes this source under Livestock GHG Emissions. The methodology outlined below is excerpted from Annex N of the U.S. GHG Inventory with permission from EPA to detail the methodology for emission estimates for Agricultural Soil Management; it does not include the portion on N<sub>2</sub>O emissions from livestock manure deposited on pasture, range, and paddock. Methods for this source are covered in Chapter 2 of this report.

N<sub>2</sub>O emissions were derived using activity data and emissions coefficients applied to the activity data. Activity data are developed for each source of nitrogen, estimated in terms of total amounts of nitrogen added. For histosol cultivation, activity data are annual areas of histosol soils that are cultivated<sup>12</sup> (Table 3-5). The activity data are derived from statistics, such as fertilizer consumption data or livestock population data, and various factors used to convert these statistics to annual amounts of nitrogen, such as fertilizer nitrogen contents or livestock excretion rates.

Nitrogen additions to soils result in direct and indirect emissions. Indirect emissions account

<sup>12</sup> Histosols are soils with a high organic carbon content. All soils with more than 20 to 30 percent organic matter by weight (depending on the clay content) are classified as histosols (Brady and Weil 1999).

for nitrogen that volatilizes to the atmosphere as NH<sub>3</sub> and NO<sub>x</sub>, and subsequently returns to soils through atmospheric deposition, enhancing N<sub>2</sub>O production. Additional nitrogen is lost from soils through leaching and runoff, and enters groundwater and surface water systems, from which a portion is emitted as N<sub>2</sub>O. These two indirect emission pathways are treated separately, although the activity data used, except for livestock manure, are identical. The activity data for livestock manure are different from those used in other calculations. Here, total livestock manure (i.e., the sum of applied manure and manure used as a livestock feed supplement) is used in the volatilization and deposition calculation; and livestock manure applied on soils (i.e., applied manure) in the leaching and runoff calculation. Indirect emissions from manure deposited in pasture, range, and paddock are addressed in Chapter 2.

### 3.10.1 Activity Data

The activity data for this component include: a) the amount of nitrogen in synthetic and organic commercial fertilizers that are applied annually, b) the amount of nitrogen in livestock manure that is applied annually through both daily spread operations and the eventual application of manure that had been stored in manure management systems<sup>13</sup>, c) the amount of nitrogen in sewage sludge that is applied annually, d) the amount of nitrogen in the aboveground biomass of nitrogen-fixing crops that are produced annually, e) the amount of nitrogen in crop residues that are retained on soils annually, and f) the area of histosols cultivated annually.

### 3.10.2 Application of Synthetic and Organic Commercial Fertilizer

Annual commercial fertilizer consumption data for the United States were taken from annual publications of synthetic and organic fertilizer statistics (TVA 1991, 1992a, 1993, 1994; AAPFCO 1995, 1996, 1997, 1998, 1999, 2000b, 2002) and a recent Association of American Plant Food Control Officials (AAPFCO) database (AAPFCO 2000a). These data were manipulated in several ways to derive the activity data needed for the inventory. First, the manure and sewage sludge portions of the organic fertilizers were subtracted from the total organic fertilizer consumption data because these nitrogen additions are accounted for under

**Table 3-5 Cultivated histosol areas**

	Temperate	Sub-tropical
	<i>1,000 ha</i>	
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

Source: USDA NRCS (2000); EPA (2003a)

<sup>13</sup> This source is distinguished from livestock manure deposited on pasture, range, and paddock because of the intent of the manure application and the land use resulting in the emission. In this case, manure has been collected and stored in a livestock operation. Emissions caused by the manure while in a storage system are accounted for in Chapter 2 and are attributed to livestock operations. After manure is removed from the storage system and used as an amendment for crop production, the resulting emissions are covered under this section on cropland soils.

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“manure application” and “sewage sludge application.”<sup>14</sup> Second, the organic fertilizer data, which are recorded in mass units of fertilizer, had to be converted to mass units of nitrogen by multiplying by the average organic fertilizer nitrogen contents provided in the annual fertilizer publications. These nitrogen contents are weighted average values, so they vary from year to year (ranging from 2.3 percent to 3.9 percent over the period 1990 through 2001). The synthetic fertilizer data are recorded in units of nitrogen, so these data did not need to be converted. Lastly, both the synthetic and organic fertilizer consumption data are recorded in “fertilizer year” totals (i.e., July to June); therefore, the data were converted to calendar year totals. This was done by assuming that approximately 35 percent of fertilizer usage occurred from July to December, and 65 percent from January to June (TVA 1992b). July to December values were not available for calendar year 2001, so a “least squares line” statistical extrapolation using the previous 11 years of data was used to arrive at an approximate value. Annual consumption of commercial fertilizers synthetic and non-manure/non-sewage organic in units of nitrogen and on a calendar year basis are presented in Figure 3-7 (A) and (C).

### 3.10.3 Application of Livestock Manure

To estimate the amount of livestock manure nitrogen applied to soils, it was assumed that all of the manure produced by livestock would be applied to soils with two exceptions: (1) the portion of poultry manure that is used as a feed supplement for ruminants, and (2) the manure that is deposited on soils by livestock on pasture, range, and paddock. In other words, it is assumed that all of the managed manure, except the portion of poultry manure that is used as a feed supplement, is applied to soils. The amount of managed manure for each livestock type was calculated by determining the population of animals that were on feedlots or otherwise housed in order to collect and manage the manure. In some instances, the number of animals in managed systems was determined by subtracting the number of animals in pasture, range, and paddock from the total animal population for a particular animal type. Annual animal population data for all livestock types, except horses and goats, were obtained for all years from the USDA NASS (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000 ; Hogs and Pigs: 1998, 1994; Chicken and Eggs: 1998; Poultry Production and Value: 1999, 1995; Sheep and Goats: 1999 1994). Horse population data were obtained from the FAOSTAT database (FAO 2002). Goat population data for 1992 and 1997 were obtained from the Census of Agriculture (USDA NASS Census 1999); these data were interpolated and extrapolated to derive estimates for the other years. Information regarding poultry turnover (i.e., slaughter) rate was obtained from State NRCS personnel (Lange 2000). Additional population data for different farm size categories for dairy and swine were obtained from the Census of Agriculture (USDA NASS Census 1999). Information regarding the percentage of manure handled using various manure management systems for dairy cattle, beef cattle, and sheep was obtained from communications with personnel from State NRCS offices, State universities, NASS, and other

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<sup>14</sup> Organic fertilizers included in these publications are manure, compost, dried blood, sewage sludge, tankage, and “other.” (Tankage is dried animal residue, usually freed from fat and gelatin). The manure and sewage sludge used as commercial fertilizer are accounted for elsewhere, so these were subtracted from the organic fertilizer statistics to avoid double counting.

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experts (Poe et al. 1999, Anderson 2000, Deal 2000, Johnson 2000, Miller 2000, Milton 2000, Stettler 2000, Sweeten 2000, Wright 2000). Information regarding the percentage of manure handled using various manure management systems for swine, poultry, goats, and horses was obtained from Safley et al. (1992). A more detailed discussion of manure management system usage is provided in Chapter 2 of this report and Annex M of the U.S. GHG Inventory.

Once the animal populations for each livestock type and management system were estimated, populations were multiplied by an average animal mass constant (USDA NRCS 1996, USDA NRCS 1998, ASAE 1999, Safley 2000) to derive total animal mass for each animal type in each management system. Total Kjeldahl nitrogen<sup>15</sup> excreted per year for each livestock type and management system was then calculated using daily rates of nitrogen excretion per unit of animal mass (USDA NRCS 1996, ASAE 1999). The total poultry manure nitrogen in managed systems was reduced by the amount assumed used as a feed supplement (i.e., 4.2 percent of the managed poultry manure; Carpenter 1992). The annual amounts of Kjeldahl nitrogen were then summed over all livestock types and management systems to derive estimates of the annual manure nitrogen applied to soils and are shown in Figure 3-7 (B).

#### 3.10.4 Application of Sewage Sludge

Estimates of annual nitrogen additions from land application of sewage sludge were derived from periodic estimates of sludge generation and disposal rates that were developed by EPA. Sewage sludge is generated from the treatment of raw sewage in public or private wastewater treatment works. Based on a 1988 questionnaire returned from 600 publicly owned treatment works (POTWs), the EPA estimated that 5.4 million metric tons of dry sewage sludge were generated by POTWs in the United States in that year (EPA 1993a). Of this total, 33.3 percent was applied to land, including agricultural applications, compost manufacture, forestland application, and the reclamation of mining areas. A subsequent EPA report (EPA, 1999) compiled data from several national studies and surveys, and estimated that approximately 6.7 and 6.9 million metric tons of dry sewage sludge were generated in 1996 and 1998, respectively, from all treatment works, and projected that approximately 7.1 million metric tons would be generated in 2000. The same study concluded that 60 percent of the sewage sludge generated in 1998 was applied to land (based on the results of a 1995 survey), and projected that 63 percent would be land applied in 2000. These EPA estimates of sludge generation and percent applied to land were linearly interpolated to derive estimates for each year in the 1990-2000 period. To estimate annual amounts of nitrogen applied, the annual amounts of dry sewage sludge applied were multiplied by an average nitrogen content of 3.3 percent (Metcalf and Eddy, Inc. 1991). Estimates for the year 2001 were held constant at the year 2000 level, as no new data were available (Bastian, 2002). Final estimates of annual amounts of sewage sludge nitrogen applied to land are presented in Figure 3-7 (C).

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<sup>15</sup> Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen in both the solid and liquid wastes.

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### 3.10.5 Production of Nitrogen-Fixing Crops

Annual production statistics for beans, pulses, and alfalfa were taken from crop production reports (USDA NASS: Field Crops 1994 and 1998; Crop Production 2000, 2001, and 2002). Annual production statistics for the remaining nitrogen-fixing crops (i.e., the major non-alfalfa forage crops, specifically red clover, white clover, birdsfoot trefoil, arrowleaf clover, and crimson clover) were derived from information in a book on forage crops (Taylor and Smith 1995, Pederson 1995, Beuselinck and Grant 1995, Hoveland and Evers 1995), and personal communications with forage experts (Cropper 2000, Evers 2000, Gerrish 2000, Hoveland 2000, and Pederson 2000). The production statistics for beans, pulses, and alfalfa were in tons of product, which needed to be converted to tons of aboveground biomass nitrogen. This was done by multiplying the production statistics by 1 plus the aboveground residue-to-crop product mass ratios, dry matter fractions, and nitrogen contents. The residue to crop product mass ratios for soybeans and peanuts and the dry matter content for soybeans, were obtained from Strehler and Stützle (1987). The dry matter content for peanuts was obtained through personal communications with Ketzis (1999). The residue-to-crop product ratios and dry matter contents for the other beans and pulses were estimated by taking averages of the values for soybeans and peanuts. The dry matter content for alfalfa was obtained through personal communications with Karkosh (2000). The IPCC default nitrogen content of 3 percent (IPCC/UNEP/OECD/IEA 1997) was used for all beans, pulses, and alfalfa.<sup>16</sup> The production statistics for the non-alfalfa forage crops were derived by multiplying estimates of areas planted by estimates of annual yields, in dry matter mass units. These derived production statistics were then converted to units of nitrogen by applying the IPCC default nitrogen content of 3 percent (IPCC/UNEP/OECD/IEA 1997). The final estimates of annual aboveground biomass production, in units of nitrogen, are in Appendix Table B-9. The residue to crop product mass ratios and dry matter fractions used in these calculations are presented in Appendix Table B-8.

### 3.10.6 Retention of Crop Residue

It was assumed that 90 percent of residues from corn, wheat, barley, sorghum, oats, rye, millet, soybeans, peanuts, and other beans and pulses are left on the field after harvest (e.g., rolled into the soil, chopped and disked into the soil, or otherwise left behind) (Karkosh 2000).<sup>17</sup> It was also assumed that 100 percent of unburned rice residue is left on the field.<sup>18</sup> The derivation of residue nitrogen activity data was very similar to the derivation of nitrogen-fixing crop activity data. Crop production statistics were multiplied by aboveground residue to crop product mass

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<sup>16</sup> This nitrogen content may be an overestimate for the residue portion of the aboveground biomass of the beans and pulses. Also, the dry matter fractions used for beans and pulses were taken from literature on crop residues, and so may be underestimates for the product portion of the aboveground biomass.

<sup>17</sup> Although the mode of residue application would likely affect the magnitude of N<sub>2</sub>O emissions, an emission estimation methodology that accounts for this has not been developed.

<sup>18</sup> Some of the rice residue may be used for other purposes, such as for biofuel or livestock bedding material. Research to obtain more detailed information regarding final disposition of rice residue, as well as the residue of other crops, will be undertaken for future inventories.

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ratios, residue dry matter fractions, residue nitrogen contents, and the fraction of residues left on soils. Annual production statistics for all crops except rice in Florida were taken from USDA NASS (Field Crops 1994 and 1998; Crop Production: 2001 and 2002). Production statistics for rice in Florida were estimated by applying an average rice crop yield for Florida (Smith 2001) to annual Florida rice acreages (Schueneman 1999, 2001, Deren 2002).

Residue to crop product ratios for all crops were obtained from, or derived from, Strehler and Stütze (1987). Dry matter contents for wheat, rice, corn, and barley residue were obtained from Turn et al. (1997). Soybean and millet residue dry matter contents were obtained from Strehler and Stütze (1987). Peanut, sorghum, oat, and rye residue dry matter contents were obtained through personal communications with Ketzis (1999). Dry matter contents for all other beans and pulses were estimated by averaging the values for soybeans and peanuts. The residue nitrogen contents for wheat, rice, corn, and barley are from Turn et al. (1997). The nitrogen content of soybean residue is from Barnard and Kristoferson (1985), the nitrogen contents of peanut, sorghum, oat, and rye residue are from Ketzis (1999), and the nitrogen content of millet residue is from Strehler and Stütze (1987). Nitrogen contents of all other beans and pulses were estimated by averaging the values for soybeans and peanuts. Estimates of the amounts of rice residue burned annually were derived using information obtained from agricultural extension agents in each of the rice-growing States (see methods for Agricultural Residue Burning for more detail). The final estimates of residue retained on soil, in units of nitrogen (N), are in Appendix Table B-10. The residue to crop product mass ratios, residue dry matter fractions, and residue nitrogen contents used in these calculations are in Appendix Table B-8.

### 3.10.7 Cultivation of Histosols

Estimates of the areas of histosols cultivated in 1982, 1992, and 1997 were obtained from the USDA's 1997 National Resources Inventory (USDA NRCS 2000, as extracted by Eve 2001, and revised by Ogle 2002). These areas were grouped by broad climatic region using temperature and precipitation estimates from Daly et al. (1994, 1998), and then further aggregated to derive a temperate total and a sub-tropical total. These final areas were then linearly interpolated to obtain estimates for 1990 through 1996, and linearly extrapolated to obtain area estimates for 1998 through 2001 (Table 3-5).

### 3.10.8 Direct N<sub>2</sub>O Emissions From Cropland Soils

Direct N<sub>2</sub>O emissions from nitrogen additions and histosol cultivation were calculated separately, and each approach is discussed below. To estimate these direct emissions from nitrogen additions, the amounts of nitrogen applied were each reduced by IPCC values for fraction of nitrogen that is assumed to volatilize, the unvolatilized amounts were then summed, and the total unvolatilized nitrogen was multiplied by an emission factor of 0.0125 kg N<sub>2</sub>O-N/kg Nitrogen (IPCC/UNEP/OECD/IEA 1997). The volatilization assumptions inherent in this approach are described below:

- Application of synthetic and organic commercial fertilizer: The total amounts of nitrogen applied in the form of synthetic commercial fertilizers and non-manure/non-sewage organic commercial fertilizers were reduced by 10 percent and 20 percent, respectively, to account for the portion that volatilizes to NH<sub>3</sub> and NO<sub>x</sub> (IPCC/UNEP/OECD/IEA 1997).
- Application of livestock manure: The total amount of livestock manure nitrogen applied to soils was reduced by 20 percent to account for the portion that volatilizes to NH<sub>3</sub> and NO<sub>x</sub> (IPCC/UNEP/OECD/IEA 1997).
- Application of sewage sludge: The total amount of sewage sludge nitrogen applied to soils was reduced by 20 percent to account for the portion that volatilizes to NH<sub>3</sub> and NO<sub>x</sub> (IPCC/UNEP/OECD/IEA 1997, IPCC 2000).
- Production of nitrogen-fixing crops: None of the nitrogen in the aboveground biomass of nitrogen-fixing crops was assumed to volatilize.
- Retention of crop residue: None of the nitrogen in retained crop residue was assumed to volatilize.

To estimate annual N<sub>2</sub>O emissions from histosol cultivation, the temperate histosol area was multiplied by the IPCC default emission factor for temperate soils (8 kg N<sub>2</sub>O-N/ha cultivated; IPCC 2000), and the sub-tropical histosol area was multiplied by the average of the temperate and tropical IPCC default emission factors (12 kg N<sub>2</sub>O-N/ha cultivated; IPCC 2000).

### 3.10.9 Indirect N<sub>2</sub>O Emissions Induced by Applications of Nitrogen

To estimate N<sub>2</sub>O emission from volatilization, the amounts of commercial fertilizer nitrogen and sewage sludge nitrogen applied, and the total amount of manure nitrogen produced, were each multiplied by the IPCC default fraction of nitrogen that is assumed to volatilize to NH<sub>3</sub> and NO<sub>x</sub> (10 percent for synthetic fertilizer nitrogen; and 20 percent for nitrogen in organic fertilizer, sewage sludge, and livestock manure). Next, the volatilized amounts of nitrogen were summed, and then the total volatilized nitrogen was multiplied by the IPCC default emission factor of 0.01 kg N<sub>2</sub>O-N/kg N (IPCC/UNEP/OECD/IEA 1997).

To estimate indirect emissions from leaching and runoff, the amounts of commercial fertilizer nitrogen and sewage sludge nitrogen applied, and the total amount of manure nitrogen applied, were each multiplied by the IPCC default fraction of nitrogen that is assumed to leach and runoff (30 percent for all nitrogen). Next, the leached/runoff amounts of nitrogen were summed, and then the total nitrogen was multiplied by the IPCC default emission factor of 0.025 kg N<sub>2</sub>O-N/kg K (IPCC/UNEP/OECD/IEA 1997).

## 3.11 Uncertainty in Estimating N<sub>2</sub>O Emissions from Agricultural Soils

The following discussion of uncertainty in estimating GHG emissions from cropland soils is modified from information in the U.S. GHG Inventory and reproduced here with permission from EPA.

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The magnitude of N<sub>2</sub>O emissions from cropland soils depends on nitrogen inputs and soil characteristics such as organic carbon availability, O<sub>2</sub> partial pressure, soil moisture content, pH, soil temperature, and soil amendments. The interacting impact of these variables on N<sub>2</sub>O flux is complex and highly uncertain. Therefore, the IPCC default methodology, which is used here, is based only on N inputs and does not consider soil characteristics. This generalized approach treats all soils, except cultivated histosols, the same.

IPCC default emission factors for N<sub>2</sub>O have associated ranges, the magnitudes of which indicate the uncertainty in the emission estimates. Most emission factor ranges are an order of magnitude, or larger. Developing a method to explicitly consider all driving factors of N<sub>2</sub>O emissions will require more research; a prototype approach is described in Box 3-1, making use of process models to estimate N<sub>2</sub>O emissions.

Uncertainties derive from activity data used to derive emission estimates. Activity data were often extrapolated from periodic surveys and commercial sales data, or, in some cases, they were based on expert opinion. For example: fertilizer statistics used in these estimates only include organic fertilizers in the commercial market; livestock excretion values were derived using simplifying assumptions concerning the types of management systems employed; and annual production and application estimates for sewage sludge were based on figures and projections calculated from surveys, yielding uncertainty levels as high as 14 percent (Bastian 1999). In addition, expert judgment was used to estimate the amount of residues left on soils, and the area of cultivated histosol soils was extrapolated from periodic data collected by a natural resource inventory, which was not explicitly designed as a soil survey. Production statistics for nitrogen-fixing forage legumes are not available except in the case of alfalfa; even so the statistics include alfalfa mixtures. Finally, conversion factors for nitrogen-fixing crops are based on a limited number of studies.

## 3.12 Mitigating N<sub>2</sub>O and CH<sub>4</sub> Emissions from Cropland Agriculture

### 3.12.1 N<sub>2</sub>O From Agricultural Soils

Fertilizer nitrogen (N) use efficiency in agricultural systems is limited by large losses of N through leaching and transformation to gaseous forms of N-oxides (ammonia, N<sub>2</sub>O, nitric oxide, dinitrogen). In general, N-oxide emissions from mineral and organic soils can be reduced by management practices optimizing crop use of available N and minimizing losses from the soil-plant system. Strategies to increase overall N efficiency can consequently decrease N-oxide production, including N<sub>2</sub>O emissions (Mosier et al. 1998a).

Sixty-nine percent of N<sub>2</sub>O emissions from cropland soils are direct emissions, while the remaining 31 percent are emitted indirectly through volatilization and runoff. Management options focus on the direct sources of emissions; however, decreases in external N additions to crops will decrease both direct and indirect N<sub>2</sub>O emissions. Current research, development, and

deployment efforts in the area of nutrient management focus on the following areas:

- *Precision agriculture* – targeted application of fertilizers, water, and pesticides.
- *Cropping system models* - tools to assist farm management decisions.
- *Controlled release of fertilizers and pesticides* – delivery of nutrients and chemicals to match crop demand and timing of pest infestation.
- *Soil microbial processes* – use of biological and chemical methods to manipulate microbial processes to increase efficiency of nutrient uptake, suppress N<sub>2</sub>O emissions, and reduce leaching.
- *Agricultural best management practices* - limit N-gas emissions, soil erosion, and leaching.
- *Soil conservation practices* - utilizing conservation buffers and reserves.
- *Livestock manure utilization* - development of mechanisms to more effectively use livestock manure in crop production.
- *Plant breeding* – breeding varieties to increase nutrient use efficiency and decrease demand for pesticides, thus conserving energy.

### 3.12.2 CH<sub>4</sub> Emissions From Rice Fields

The amount of CH<sub>4</sub> emissions depends primarily on the area of rice cultivation. Since 1996, rice acreage has been controlled mainly by spring commodity prices and weather. Continuously flooded rice production is the most common production technique in the United States. Given the comparatively low level of emissions from rice paddies in the United States, there are no programs directed at limiting CH<sub>4</sub> emissions from rice fields. However, available research in the United States and internationally suggests that the most viable CH<sub>4</sub> reduction methods are cultivar selection, water management, and nutrient management.

### 3.13 Carbon Stock Changes in Agricultural Soils

**Table 3-6 Net carbon dioxide flux from agricultural soils, 1990, 1995-2001**

	1990	1995	1996	1997	1998	1999	2000	2001
	<i>Tg CO<sub>2</sub> eq.</i>							
Mineral Soils	(57.1)	(58.6)	(57.3)	(57.4)	(55.8)	(55.7)	(57.3)	(59.1)
Organic Soils	34.3	34.8	34.8	34.8	34.8	34.8	34.8	34.8
Liming of Soils	9.5	8.9	8.9	8.7	9.6	9.1	8.8	9.1
Total	(13.3)	(14.9)	(13.6)	(13.9)	(11.5)	(11.9)	(13.8)	(15.2)

Note: Parentheses indicate net sequestration. Shaded values based on a combination of historical data and projections. All other values are based on historical data only.

<sup>19</sup> Emissions and sinks of carbon in agricultural soils are expressed in terms of CO<sub>2</sub> equivalents; carbon sequestration is a result of changes in stocks of carbon in soils, from which CO<sub>2</sub> fluxes are inferred. Units of CO<sub>2</sub> equivalent can be converted to carbon using a multiplier of 0.272.

Except for cultivated organic soils and liming practices, agricultural soils in the United States were estimated to accumulate about 59 Tg CO<sub>2</sub> eq. in 2001 (Table 3-6).<sup>19</sup> 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. 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.

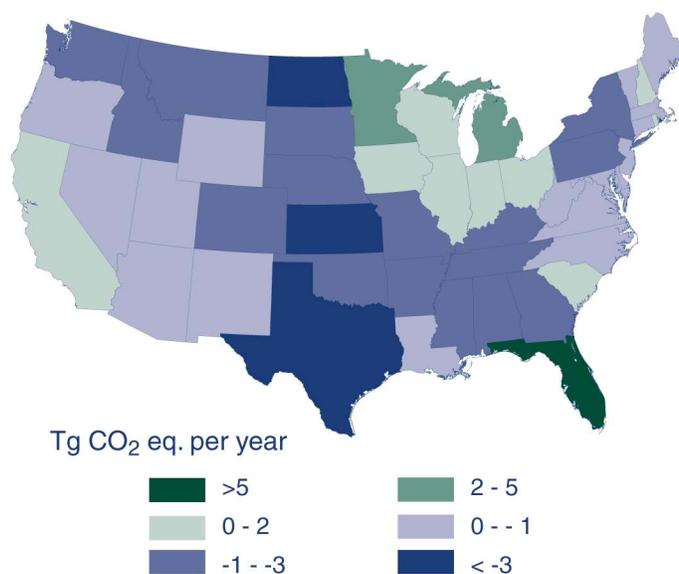
In contrast, the small area of cultivated organic soils—less than 1 million hectares of a total 386 million hectares of agricultural land—concentrated in Florida, California, the Gulf and Southeastern coastal region and parts of the upper Midwest, was a net source of CO<sub>2</sub> emissions for all years covered by the inventory (1990-2001). About 35 Tg CO<sub>2</sub> eq. was emitted from cultivation of these soils in 2001 (Table 3-6). In addition, liming of agricultural soils resulted in emissions of about 9 Tg CO<sub>2</sub> eq. Total net carbon sequestration in 2001 was about 15 Tg CO<sub>2</sub> eq. when all of the above components are taken into consideration.

Carbon uptake on agricultural soils increased 14 percent (1.9 Tg CO<sub>2</sub> eq.) between 1990 and 2001 (Table 3-6). This trend is a function of increases in carbon uptake on mineral soils and decreases in carbon losses from liming, with slight increases in carbon losses on organic soils. Carbon uptake in mineral soils increased by 2 Tg CO<sub>2</sub> eq (3.5 percent), emissions from liming decreased slightly (<0.4 percent), and carbon losses on organic soils increased by less than 1 Tg CO<sub>2</sub> eq. (1.5 percent).

The remainder of this section focuses on carbon stock changes in mineral and organic soils. National estimates of carbon stock changes in mineral and organic agricultural soils are derived from several data sources including the Natural Resources Inventory (NRI), which is conducted every 5 years (USDA NRCS 2000). NRI data from 1982 through 1997 were used to derive carbon stock changes resulting from transitions in land use and management occurring over this time period.

State-level estimates were developed to provide a finer scale of information on trends in agricultural carbon stock changes. The national estimates for 1997 were disaggregated at the

Map 3-1  
CO<sub>2</sub> emissions and sequestration in agricultural soils, 1997



Note: Negative values correspond to sequestration.

**Table 3-7 Areas in each land-use and management system for all U.S. land areas categorized as an agricultural use in 1992 or 1997 in the NRI**

IPCC Land Use/Management Categories	1982	1992	1997
	<i>Million hectares</i>		
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 (hayland, 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
<b>Total</b>	<b>386.39</b>	<b>386.39</b>	<b>386.39</b>

Source: USDA NRCS (2000).

is the most recent year for which NRI data are available. For example, all lands that were in CRP as of 1997 are grouped together, irrespective of their prior management in earlier NRI data sets. Therefore, changes in aggregate carbon stocks for land classified as CRP in 1997 are a function of all changes in cropping system/land use that occurred after 1982, including the most recent change into CRP.

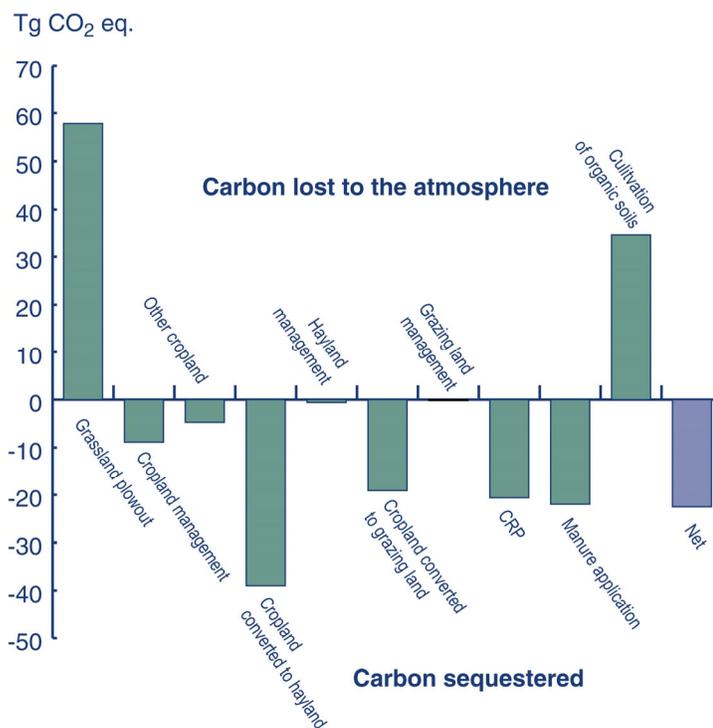
While total areas of annual cropland, hayland, and pasture are relatively constant over the inventory period, the NRI data indicate a substantial area of hayland and pasture was converted to annual cropping between 1982 and 1997 and a roughly corresponding area of annual cropland was put into hay land and pasture (Table 3-7). To better illustrate the effects of this land use “switching,” data are provided separately for cropland and hayland/pasture, showing the gross effects of the land use change on soil carbon stocks, along with changes in carbon stocks occurring on land areas that underwent this “switching” of land use during the inventory period.

level of Major Land Resource Area (MLRA) and used to derive an approximate State-level estimate of carbon stock changes. State totals were computed using the values for all MLRAs occurring in the State, weighted by the share of the total State area comprised by each MLRA. Underlying trends in carbon stock changes are discussed below, focusing on individual components of the net soil carbon estimates (e.g., land set aside under CRP, manure applications, tilling practices, land use conversions between annual and perennial systems) and providing information on the magnitude of each component at the State level.

Given the complexity of illustrating the full range of carbon stock changes associated with each transition over the entire time sequence, the State-level analysis presented below shows net changes in agricultural soil carbon according to the major land use or management categories to which they belonged in 1997, which

Net carbon stock changes in agricultural soils are shown for each State in Map 3-1 for 1997. These net changes are comprised of the sum of changes in individual components both in positive (carbon uptake) and negative (carbon losses to the atmosphere) directions (Appendix Table B-11). In 1997, crop cultivation resulted in carbon losses to the atmosphere on the order of 93 Tg CO<sub>2</sub> eq., driven mainly by losses following conversion of pasture and hayland into annual cropping systems (emissions of 58 Tg CO<sub>2</sub> eq.) and by losses from the cultivation of carbon-rich organic soils (emissions of 35 Tg CO<sub>2</sub> eq.) (Figure 3-8). However, these losses were more than offset by land use conversions from cultivated cropland to hay and grazing lands, which sequestered close to 60 Tg CO<sub>2</sub> eq. of carbon in soils. Lands enrolled in the Conservation Reserve Program and the application of manure fertilizer to crop and grazing lands provided further carbon uptake in soils of about 40 Tg CO<sub>2</sub> eq. The net effect of these land management actions and land use conversions is the sequestration of about 23 Tg CO<sub>2</sub> eq. of carbon in agricultural soils.

Figure 3-8  
**Components of CO<sub>2</sub> emissions and sequestration in agricultural soils, 2001**



Land set aside from production into the Conservation Reserve Program is an important contributor to soil carbon stock increases (Figure 3-8), with the highest amounts registered for States where there are large areas of CRP enrollment (e.g., North Dakota, South Dakota, Texas, Kansas, Montana, Colorado) (Appendix Table B-11). Most States show small gains on long-term annual cropland, due to increased use of more carbon-conserving practices such as conservation tillage (0.15 Tg CO<sub>2</sub> eq. in Colorado; 0.48 Tg CO<sub>2</sub> eq. in Illinois; 0.66 Tg CO<sub>2</sub> eq. in Nebraska; 0.59 Tg CO<sub>2</sub> eq. in Texas) (Appendix Table B-11). Manure additions at the State level contributed to carbon sequestration in soils of 0.5 Tg CO<sub>2</sub> eq. on average, with State levels ranging from no impact of manure application to sequestration levels of nearly 2 Tg CO<sub>2</sub> eq. The largest gains from manure application were in the major livestock-producing States (e.g., California, Texas, Wisconsin, Iowa, North Carolina, Nebraska) (Appendix Table B-11). CO<sub>2</sub> emissions from cultivated organic soils are restricted to a few States having significant areas of

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those soils. Nearly one-third of the national total occurs in Florida. Other major States include California and several States in the Great Lakes region (e.g., Minnesota, Michigan, Wisconsin, and Indiana).

### 3.14 Methods for Estimating Carbon Stock Changes in Agricultural Soils

Researchers at the Natural Resources Ecology Laboratory (NREL) at Colorado State University developed the methods and provided estimates for this chapter (Ogle et al. 2003). The estimates were originally developed for the U.S. GHG Inventory following an approach modified from the Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories (IPCC/UNEP/OECD/IEA 1997). The approach was expanded for this report to include State-level estimates of carbon stock changes. The analysis was designed first and foremost to inform policymakers at the national level, using information derived from the U.S. national-level greenhouse gas inventory. The objectives of this report are to provide a preliminary assessment of changes in soil carbon emissions/sinks at the State level.

#### 3.14.1 Stock Change Calculations for Mineral Soils

For mineral soils, the method estimates effects of land use change and management on relative stock changes over a defined time interval (default of 20 years) and soil depth (default of 30 cm). The method assumes that, in the absence of any changes in management or land use over the 20-year inventory period, soil carbon stocks do not change. In essence, they are assumed to be in an equilibrium or steady-state condition. There are three main kinds of information necessary to apply the method: 1) stock change factors for specific land use and management practices, 2) reference carbon stocks for the baseline agricultural condition (i.e., conventionally tilled cropland with row-cropping rotations) to which the stock change factors are applied, and 3) activity data that record the changes in land use and management over time. These are combined in the following way:

$$\Delta C = [ \sum (SC_{it} - SC_{i(t-20)}) * LA_i ] / 20$$

$$SC_{it} = SC_R * F_{LU} * F_T * F_I$$

where  $SC_i$  is soil organic carbon stock for the  $i$ th parcel of land at time  $t$  and  $t-20$  years,  $LA_i$  is land area of each parcel,  $SC_R$  is the reference carbon stock and  $F_{LU}$ ,  $F_T$ ,  $F_I$  are stock change factors (for land-use type, tillage regime, and carbon input level, respectively), which define the land use and management conditions on each parcel of land. Land area parcels represent the areas associated with each type of land use/management system (as defined by the stock change factors), stratified by climate and soil type.

#### 3.14.2 Climate Zones and Reference Soil Carbon Stocks

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,

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precipitation, and potential evapotranspiration. Six of these occur in the continental United States. The PRISM (Parameter-elevation Regressions on Independent Slopes Model) 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. MLRAs were chosen as the main spatial entity for the national estimates because each MLRA represents a geographic unit with relatively similar soils, climate, water resources, and land uses (NRCS 1981) and the management/land use activity data used (see below) could be readily aggregated by MLRA.

Reference soil carbon stocks ( $SC_R$ ) 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)

### 3.14.3 Land Use and Management Factor Values

Management factors (i.e.,  $F_{LU}$ ,  $F_T$ ,  $F_I$ ) representative of U.S. conditions were estimated from published studies (Ogle et al. 2003). The factors quantify the relative carbon stock change resulting from changing land use and management, including tillage practices, cropping rotation or intensification, and land conversions (including set-asides in the Conservation Reserve Program). Studies from the United States or Canada were used that met a minimum set of criteria, including reporting of SOC stocks (or information to compute stocks), depth of sampling, time, and management treatments. Factors were estimated for the effect of management practices at 20 years for the top 30 cm of the soil profile (Appendix Table B-13).

### 3.14.4 Land Use and Management Activity Data

Land use and management data were based primarily on the National Resources Inventory (NRI) (USDA NRCS 2000). 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. Among the information in the NRI are land use category, soil description and, for agricultural lands, crop type. 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-14.

Tillage practices are not included in the NRI. Thus, supplemental data were used from the Conservation Technology Information Center (CTIC 1998), which reports tillage practices by major crops and county on an annual basis (Appendix Table B-15). CTIC data do not differentiate between continuous and intermittent use of no-tillage, which is important for estimating SOC storage. Thus regional-based estimates for continuous no-tillage (defined as 5 or more years of continuous use) were derived based on consultation with a CTIC expert (D. Towery 2001, personal communication).

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Other data used to supplement the NRI included: (1) area of cropland restored to wetland and (2) manure applications to crop and grazing land. Data were available for wetland restoration under the CRP program for the Northern Prairie Pothole Region (Euliss and Gleason 2002, personal communication), including the amount of area restored and estimates of carbon stock change over time. For manure (and sludge) application, EPA compiled data on the amount of total manure produced and available for application to agricultural land (EPA 2003a). Since data on actual application rates were not available, manure was assumed to be applied according to recommended rates based on the assimilative capacity for crops (Kellogg et al. 2000). Supplemental data are available regarding the amount of cropland area receiving manure and sewage sludge for major crops in the United States (USDA ERS 2000). The percentage of fields receiving manure and sewage sludge had been estimated between 1990 and 1997 for corn, soybeans, winter wheat, cotton, and potatoes. This information was used in conjunction with the data collected from the USDA NASS Agricultural Statistics database (<http://www.nass.usda.gov:81/ipedb/>), which provides information on the amount of land planted to each crop, for estimating the cropland area receiving manure and sewage sludge. The remaining area receiving manure and sewage sludge was assumed to occur in grazing lands (calculated as the difference between the total area receiving manure and sewage sludge and the cropland area receiving manure and sewage sludge).

### 3.14.5 Carbon Flux from Organic Soils

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. Carbon loss rates (mean and variance) were estimated based on field studies in the United States and Canada stratified by the three major temperature regimes characterized in the IPCC methodology (Appendix Table B-16). However, based on the limited field studies there was no significant difference between mean rates for the warm temperate and subtropical regions. Carbon loss from cultivated organic soils is based primarily on measurements of subsidence of the land surface over time, corrected for the relative contributions of compaction and erosion, to derive an estimate of the losses due to decomposition and CO<sub>2</sub> emission. The most recent data available from cultivated organic soils in Florida were included (D. Morris, personal communication), resulting in a decrease in the estimated loss rates to about 14 tons C per hectare per year, from 20 tons C per hectare per year suggested by the IPCC. Flux rates were applied to area estimates for cultivated organic soils by climate region to estimate emissions.

### 3.15 Uncertainty in Estimating Carbon Stock Changes in Agricultural Soils

The IPCC-based methodology uses empirical models to estimate carbon stock changes in response to specific land use and management activities on agricultural lands. For estimating carbon gains it can be viewed as inherently conservative for two reasons. First, stock change factor values are statistically derived for a 20-year time period; hence, if carbon gains in response to a specific management change are sustained over a longer time period, then gains beyond 20 years will be unaccounted for in the inventory.<sup>20</sup> Second, the IPCC method deals only with discrete changes in management systems and

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<sup>20</sup> However, changes that result in decreases in C stocks, such as conversion of native vegetation or pasture to cultivated annual crops, are also derived for a 20-yr time frame, and thus losses sustained over a longer period of time would be underestimated.

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thus longer term ‘baseline’ trends in productivity and carbon inputs to soil (i.e., independent of changes in tillage or crop rotation) are not accounted for in the present methodology. For example, several authors have suggested that the general increase in U.S. agricultural productivity (on the order of 1-2 percent annually for major crop species over the past 50 years) has contributed to soil carbon sequestration through a steady increase in crop residue production (e.g., Cole et al. 1993, Allmaras et al. 2000). Because this steady productivity increase is not readily characterized as a ‘change’ in land use or management, it is difficult to capture in the IPCC method as presently implemented. Finally, the IPCC-based method is currently applied at a highly aggregated level,<sup>21</sup> and thus may not capture many regional differences in how soil carbon stocks respond to management.

Carbon loss from cultivated organic soils is based primarily on measurements of subsidence of the land surface over time, corrected for the relative contributions of compaction and erosion, to derive an estimate of the losses due to decomposition and CO<sub>2</sub> emission. Few studies are available, leading to significant uncertainty in the estimates of carbon flux rates from organic soils. In addition, some of the published data are from older studies in which loss rates may be higher than they would be at present if better water management practices (i.e., maintaining higher water tables) were used.

Annex P of the U.S. GHG Inventory (EPA 2003a) contains details on the quantitative uncertainties associated with national estimates of CO<sub>2</sub> emissions and sinks in agricultural soils.

### 3.16 Alternative Approaches for Estimating Carbon Stock Changes

Alternative approaches using dynamic simulation models such as the Century model have been used to estimate changes in agricultural soil carbon emissions and sinks at national and regional scales (e.g., Paustian et al. 2002, Falloon et al. 2002, Brenner et al. 2001, 2002, Smith et al. 2002). Such dynamic models offer some advantages in that factors such as long-term trends in crop productivity, as well as finer scale representation of climate, soil, and management conditions, can more easily be incorporated. In addition, since they operate as a continuous simulation (as opposed to a discrete time period as in the IPCC method), they are better suited to producing annual estimates.

For example, dynamic simulation models were used to study agricultural soil carbon changes in Nebraska (Brenner et al. 2002). Data on climate, soils, land use, and management were collected for each county in the State, with assistance from local conservation district and USDA NRCS personnel. These data were used as inputs to the Century model to estimate soil carbon emissions and sinks from 1990 to 2000 and to make projections of potential changes in soil carbon with increased adoption of conservation practices. For comparison, the Century model study estimated that Nebraska agricultural soils in 1997 sequestered 4.62 Tg CO<sub>2</sub> eq. per

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<sup>21</sup> Work is underway to develop a more disaggregated set of factor values to better reflect regional climate and soil influences on C stock responses to management.

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year, compared to the State-level estimate of 1.98 Tg CO<sub>2</sub> eq. based on the IPCC method.

For some practices, such as the effect of the Conservation Reserve Program on carbon stock change, estimates were similar ( 0.92 Tg CO<sub>2</sub> eq. per year using Century and 0.99 Tg CO<sub>2</sub> eq. per year using IPCC). The Century analysis estimated that much of the total carbon gains in the State occurred on annual cropland under intensive and reduced tillage (0.92 and 2.42 Tg CO<sub>2</sub> eq. respectively), which comprises about 95 percent of the annual cropland area (ca. 20 million acres). Some of this gain is due to the long-term impact of steady increases in productivity as mentioned above, along with a shift towards less intensive tillage. An additional gain of 0.37 Tg CO<sub>2</sub> eq. per year was estimated for continuous no-till cropland, which comprised less than 1 million acres, bringing the total gain on annual cropland to 3.7 Tg CO<sub>2</sub> eq. per year using the Century approach.

In contrast, the IPCC method estimated a net sequestration increase on all cropland in Nebraska (excluding CRP) of only 1.1 Tg CO<sub>2</sub> eq. per year versus the 3.7 Tg CO<sub>2</sub> eq. per year estimated using with Century. One reason for the discrepancy between the methods is that the Century analysis did not include the offsetting effects of “switching” between annual and perennial cropland (i.e., hayland, rangeland), which in the IPCC analysis accounted for a net loss of about 0.7 Tg CO<sub>2</sub> eq. per year. However, the overall increase in agricultural productivity in the United States over the past several decades likely represents some additional carbon sequestration that is not accounted for in the IPCC method. Two other State-level studies in Iowa and Indiana have been conducted using the same model-based approach. In both cases, the results were substantially higher estimates of carbon sequestration rates than in the IPCC-based approach, largely for the same reasons discussed above.

### 3.17 Summary and Recommendations

The present analysis builds on a simple method designed for national-level greenhouse gas inventories. It includes factors, estimated from U.S. field studies that encompass most of the major influences of land use and management on soil carbon stocks. The model is applied to data from the National Resources Inventory and other sources that provide a robust, statistically based estimate of agricultural management activities during the past two decades. However, the data and model components are most appropriate for use at broad scales, and thus their interpretation at sub-national levels should be viewed with caution. The State-level disaggregation presented here is intended to provide insight into possible regional trends, and allow comparison with other State-level components of agricultural greenhouse gas emissions.

Agriculture was estimated in 2001 to sequester about 59 Tg CO<sub>2</sub> eq. on mineral soils, which was partially offset by net losses of 33-37 Tg CO<sub>2</sub> eq. from cultivated organic soils, yielding a net of about 22 Tg CO<sub>2</sub> eq. (EPA 2003a). The IPCC method has also been applied to estimate *potential* soil carbon sequestration with adoption of best management practices in the United States, which yielded an estimate of about 367 Tg CO<sub>2</sub> eq. per year (Sperow et al. in 2003). This amount is similar to other independent estimates of potential soil carbon sequestration

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rates of 275-623 Tg CO<sub>2</sub> eq. per year (e.g., Bruce et al. 1999, Lal et al. 1998). The differences between these potential and actual rates are indicative of the large ‘room for improvement’ that exists in implementing agricultural practices that enhance soil carbon storage. For example, while no-till practices are becoming more widely used, long-term continuous no-till is still practiced on only a few percent of the annual crop area within most regions. Intermittent use of no-till (such as no-till soybean and more intensive tillage of corn in a soybean-corn rotation) leads to much less carbon sequestration compared to continuous no-till. Similarly, much of the positive effects of changing crop rotations to include more hay, conversions to pasture, conservation set-asides, etc., that are occurring in some locations appear to be offset to varying degrees by other changes occurring in other locations, i.e., the “switching” effect described earlier. Thus, more consistent and sustained adoption of improved practices is needed to realize the full potential for carbon sequestration.

Methodologies suitable for more local assessment and management planning to support greenhouse gas mitigation are not yet available for all parts of the United States. However, work in some States, such as described earlier for Nebraska, shows promise in making available information and decision support aids that can be applied at field and local scales. Such an approach can capture changes in carbon stocks due to long-term trends in productivity that are not tied to specific management changes and also are more applicable for annual accounting purposes. Expanding dynamic model-based estimates to other States and to national GHG accounting purposes would enable a more comprehensive accounting of all the factors influencing soil carbon stocks changes. However, applying these more complex approaches to improve carbon accounting will require additional resources and research, including robust estimates of uncertainty.

Numerous data enhancements would be beneficial to improve inventory and carbon accounting estimates. Maintaining and expanding the utility of the National Resources Inventory is crucial because the NRI represents the only statistically valid, national-level source of information on land management and land resources. Collection of additional data at NRI points, such as tillage practices, residue management, organic amendments (type and rate; e.g., manure and sewage sludge), fertilization practices, and wetland restoration, would improve the input data and reduce uncertainty in the estimates. Other potential improvements include application of remote sensing technology to estimate the area distribution and changes over time in land use, crop rotation, and tillage practices. Such data could be overlain with soil maps using a GIS, to provide ‘wall to wall’ coverage of land management changes to complement the point sampling provided by NRI. Establishing a network of soil monitoring points, where carbon stocks could be re-measured over time along with records of management practices, either in conjunction with or complementary to the NRI, would provide a valuable means for verifying estimated carbon stock changes and further improving estimation methods.