6. Land-Use Change and Forestry

This chapter provides an assessment of the net carbon dioxide (CO_2) flux[1] caused by 1) changes in forest carbon stocks, 2) changes in carbon stocks in urban trees, 3) changes in agricultural soil carbon stocks, and 4) changes in carbon stocks in landfilled yard trimmings. Seven components of forest carbon stocks are analyzed: trees, understory vegetation, forest floor, down dead wood, soils, wood products in use, and landfilled wood products. The estimated CO_2 flux from each of these forest components was derived from U.S. forest inventory data, using methodologies that are consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). Changes in carbon stocks in urban trees were estimated based on field measurements in ten U.S. cities and data on national urban tree cover, using a methodology consistent with the *Revised 1996 IPCC Guidelines*. Changes in agricultural soil carbon stocks include mineral and organic soil carbon stock changes due to use and management of cropland and grazing land, and emissions of CO_2 due to the application of crushed limestone and dolomite to agricultural soils (i.e., soil liming). The methods used to estimate all three components of changes in agricultural soil carbon stocks are consistent with the *Revised 1996 IPCC Guidelines*. Changes in yard trimming carbon stocks in landfills were estimated using analysis of life-cycle greenhouse gas emissions and sinks associated with solid waste management (EPA 1998). Note that the chapter title "Land-Use Change and Forestry" has been used here to maintain consistency with the IPCC reporting structure for national greenhouse gas inventories; however, the chapter covers land-use activities, in addition to land-use change and forestry activities. Therefore, except in table titles, the term "land use, land-use change, and forestry" will be used in the remainder of this chapter.

Unlike the assessments in other chapters, which are based on annual activity data, the flux estimates in this chapter, with the exception of those from wood products, urban trees, and liming, are based on periodic activity data in the form of forest, land-use, and municipal solid waste surveys. Carbon dioxide fluxes from forest carbon stocks (except the wood product components) and from agricultural soils (except the liming component) are calculated on an average annual basis over five or ten year periods. The resulting annual averages are applied to years between surveys. As a result of this data structure, estimated CO_2 fluxes from forest carbon stocks (except the wood product components) and from agricultural soils (except the liming component) are constant over multi-year intervals, with large discontinuities between intervals. For the landfilled yard trimmings, periodic solid waste survey data were interpolated so that annual storage estimates could be derived. In addition, because the most recent national forest, land-use, and municipal solid waste surveys were completed for the year 1997, the estimates of CO_2 flux from forests, agricultural soils, and landfilled yard trimmings are based in part on modeled projections. Carbon dioxide flux from urban trees is based on neither annual data nor periodic survey data, but instead on data collected over the period 1990 through 1999. This flux has been applied to the entire time series.

Land use, land-use change, and forestry activities in 2001 resulted in a net sequestration of 838 Tg CO_2 Eq. (229 Tg C) (Table 6-1 and Table 6-2). This represents an offset of approximately 14 percent of total U.S. CO_2 emissions. Total land use, land-use change, and forestry net sequestration declined by approximately 22 percent between 1990 and 2001. This decline was primarily due to a decline in the rate of net carbon accumulation in forest carbon stocks. Annual carbon accumulation in landfilled yard trimmings also slowed over this period, while annual carbon accumulation in agricultural soils increased. As described above, the constant rate of carbon accumulation in urban trees is a reflection of limited underlying data (i.e., this rate represents an average for 1990 through 1999).

Table 6-1: Net CO_2 Flux from Land-Use Change and Forestry (Tg CO_2 Eq.)

Sink Category	**1990**		**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**
Forests	(982.7)		(979.0)	(979.0)	(759.0)	(751.7)	(762.7)	(755.3)	(759.0)
Urban Trees	(58.7)		(58.7)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)
Agricultural	(13.3)		(14.9)	(13.6)	(13.9)	(11.5)	(11.9)	(13.8)	(15.2)

[1] The term "flux" is used here to encompass both emissions of greenhouse gases to the atmosphere, and removal of carbon from the atmosphere. Removal of carbon from the atmosphere is also referred to as "carbon sequestration."

Soils								
Landfilled Yard Trimmings	(18.2)	(11.6)	(9.7)	(9.0)	(8.7)	(7.8)	(6.9)	(5.3)
Total	**(1,072.8)**	**(1,064.2)**	**(1,061.0)**	**(840.6)**	**(830.5)**	**(841.1)**	**(834.6)**	**(838.1)**

Note: Parentheses indicate net sequestration. Totals may not sum due to independent rounding. Shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only.

Table 6-2: Net CO_2 Flux from Land-Use Change and Forestry (Tg C)

Sink Category	1990	1995	1996	1997	1998	1999	2000	2001
Forests	(268)	(267)	(267)	(207)	(205)	(208)	(206)	(207)
Urban Trees	(16)	(16)	(16)	(16)	(16)	(16)	(16)	(16)
Agricultural Soils	(4)	(4)	(4)	(4)	(3)	(3)	(4)	(4)
Landfilled Yard Trimmings	(5)	(3)	(3)	(3)	(2)	(2)	(2)	(1)
Total	**(293)**	**(290)**	**(289)**	**(229)**	**(226)**	**(229)**	**(228)**	**(229)**

Note: 1 Tg C = 1 teragram carbon = 1 million metric tons carbon. Parentheses indicate net sequestration. Totals may not sum due to independent rounding. Shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only.

Changes in Forest Carbon Stocks

For estimating carbon flux, carbon in forest ecosystems can be divided into the following five storage pools.

- Trees, including the roots, stems, branches, and foliage of living and standing dead trees.
- Understory vegetation, including shrubs and bushes, including the roots, stems, branches, and foliage.
- Forest floor, including fine woody debris, tree litter, and humus.
- Down dead wood, including logging residue and other coarse dead wood on the ground, and stumps and roots of stumps.
- Soil, including all organic material in soil.

Carbon is continuously cycled through and among these storage pools, and between forest ecosystems and the atmosphere as a result of biological processes in forests such as growth and mortality and anthropogenic activities such as harvesting, thinning, clearing, and replanting. As trees grow, carbon is removed from the atmosphere and stored in living tree biomass. As trees age, they continue to accumulate carbon until they reach maturity, at which point they are relatively constant carbon stores. As trees die and otherwise deposit litter and debris on the forest floor, decay processes release carbon to the atmosphere and also add carbon to the soil.

However, the net change in forest carbon is not equivalent to the net flux between forests and the atmosphere because timber harvests may not always result in an immediate flux of carbon to the atmosphere. Harvesting transfers carbon from one of the forest carbon storage pools to a "product pool." Once in a product pool, the carbon is emitted over time as CO_2 if the wood product combusts or decays. The rate of emission varies considerably among different product pools. For example, if timber is harvested for energy use, combustion results in an immediate release of carbon. Conversely, if timber is harvested and subsequently used as lumber in a house, it may be many decades or even centuries before the lumber is allowed to decay and carbon is released to the atmosphere. If wood products are disposed of in landfills, the carbon contained in the wood may be released years or decades later, or may be stored permanently in the landfill.

This section of the Land-Use Change and Forestry chapter quantifies the net changes in carbon stocks in five forest carbon pools and two harvested wood pools. The net change in stocks for each pool is estimated, and then the changes in stocks are summed over all pools to estimate total net flux.

Forest carbon storage pools, and the flows between them via emissions, sequestration, and transfers, are shown in Figure 6-1. In this figure, forest carbon storage pools are represented by boxes, while flows between storage pools, and between storage pools and the atmosphere, are represented by arrows. Note that the boxes are not identical to the storage pools identified in this chapter. The storage pools identified in this chapter have been altered in this graphic to better illustrate the processes that result in transfers of carbon from one pool to another, and that result in emissions to the atmosphere.

Figure 6-1. Forest Sector Carbon Pools and Flows

Approximately 33 percent (747 million acres) of the U.S. land area is forested (Smith et al. 2001). Between 1977 and 1987, forest land declined by approximately 5.9 million acres, and between 1987 and 1997, the area increased by about 9.2 million acres. These changes in forest area represent average annual fluctuations of only about 0.1 percent. Given the low rate of change in U.S. forest land area, the major influences on the current net carbon flux from forest land are management activities and the ongoing impacts of previous land-use changes. These activities affect the net flux of carbon by altering the amount of carbon stored in forest ecosystems. For example, intensified management of forests can increase both the rate of growth and the eventual biomass density[2] of the forest, thereby increasing the uptake of carbon. Harvesting forests removes much of the aboveground carbon, but trees can grow on this area again and sequester carbon. The reversion of cropland to forest land through natural regeneration will cause increased carbon storage in biomass and soils. The net effect of both forest management and land-use change involving forests is captured in these estimates.

In the United States, improved forest management practices, the regeneration of previously cleared forest areas, and timber harvesting and use have resulted in an annual net uptake (i.e., net sequestration) of carbon during the period from 1990 through 2001. Due to improvements in U.S. agricultural productivity, the rate of forest clearing for crop cultivation and pasture slowed in the late 19th century, and by 1920 this practice had all but ceased. As farming expanded in the Midwest and West, large areas of previously cultivated land in the East were taken out of crop production, primarily between 1920 and 1950, and were allowed to revert to forests or were actively reforested. The impacts of these land-use changes are still affecting carbon fluxes from forests in the East. In addition to land-use changes in the early part of this century, carbon fluxes from Eastern forests have been affected by a trend toward managed growth on private land. Collectively, these changes have produced a near doubling of the biomass density in eastern forests since the early 1950s. More recently, the 1970s and 1980s saw a resurgence of federally sponsored forest management programs (e.g., the Forestry Incentive Program) and soil conservation programs (e.g., the Conservation Reserve Program), which have focused on tree planting, improving timber management activities, combating soil erosion, and converting marginal cropland to forests. In addition to forest regeneration and management, forest harvests have also affected net carbon fluxes. Because most of the timber that is harvested from U.S. forests is used in wood products and much of the discarded wood products are disposed of by landfilling, rather than incineration, significant quantities of this harvested carbon are transferred to long-term storage pools rather than being released to the atmosphere. The size of these long-term carbon storage pools has also increased over the last century.

Changes in carbon stocks in U.S. forests and harvested wood were estimated to account for an average annual net sequestration of 887 Tg CO_2 Eq. (242 Tg C) over the period 1990 through 2001 (Table 6-3 and Table 6-4).[3] The net sequestration is a reflection of net forest growth and increasing forest area over this period, particularly from 1987 to 1997, as well as net accumulation of carbon in harvested wood pools. The rate of annual sequestration, however, declined by 23 percent between 1990 and 2001. This was due to a greater increase in forest area between 1987 and 1997 than between 1997 and 2001. Most of the decline in annual sequestration occurred in the forest soil carbon pool. This result is due to the method used to account for changes in soil carbon after the conversion of land from forest to non-forest. Specifically, soil carbon stocks for each forest type are assumed to depend on land use

[2] The term "biomass density" refers to the weight of vegetation per unit area. It is usually measured on a dry-weight basis. Dry biomass is about 50 percent carbon by weight.

[3] This average annual net sequestration is based on the entire time series (1990 through 2001), rather than the abbreviated time series presented in Table 6-3 and Table 6-4. Results for the entire time series are presented in Annex O.

and soil type and not to vary over time within forests. Therefore, as lands are converted from non-forest to forest, there is a substantial immediate increase in soil carbon stocks.

Table 6-3: Net Changes in Carbon Stocks in Forest and Harvested Wood Pools, and Total Net Forest Carbon Flux (Tg CO_2 Eq.)

Carbon Pool	1990		1995	1996	1997	1998	1999	2000	2001
Forest	**(773.7)**		**(773.7)**	**(773.7)**	**(546.3)**	**(546.3)**	**(546.3)**	**(546.3)**	**(546.3)**
Trees	(469.3)		(469.3)	(469.3)	(447.3)	(447.3)	(447.3)	(447.3)	(447.3)
Understory	(11.0)		(11.0)	(11.0)	(14.7)	(14.7)	(14.7)	(14.7)	(14.7)
Forest Floor	(25.7)		(25.7)	(25.7)	29.3	29.3	29.3	29.3	29.3
Down Dead Wood	(55.0)		(55.0)	(55.0)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)
Forest Soils	(212.7)		(212.7)	(212.7)	(55.0)	(55.0)	(55.0)	(55.0)	(55.0)
Harvested Wood	**(209.0)**		**(205.3)**	**(205.3)**	**(212.7)**	**(205.3)**	**(216.3)**	**(209.0)**	**(212.7)**
Wood Products	(47.7)		(55.0)	(55.0)	(58.7)	(51.3)	(62.3)	(58.7)	(58.7)
Landfilled Wood	(161.3)		(150.3)	(150.3)	(154.0)	(154.0)	(154.0)	(150.3)	(154.0)
Total Net Flux	**(982.7)**		**(979.0)**	**(979.0)**	**(759.0)**	**(751.7)**	**(762.7)**	**(755.3)**	**(759.0)**

Note: Parentheses indicate net carbon "sequestration" (i.e., accumulation into the carbon pool minus emissions or stock removal from the carbon pool). The sum of the net stock changes in this table (i.e., total net flux) is an estimate of the actual net flux between the total forest carbon pool and the atmosphere. Shaded areas indicate values based on a combination of historical data and projections. Forest values are based on periodic measurements; harvested wood estimates are based on annual surveys. Totals may not sum due to independent rounding.

Table 6-4: Net Changes in Carbon Stocks in Forest and Harvested Wood Pools, and Total Net Forest Carbon Flux (Tg C)

Carbon Pool	1990		1995	1996	1997	1998	1999	2000	2001
Forest	**(211)**		**(211)**	**(211)**	**(149)**	**(149)**	**(149)**	**(149)**	**(149)**
Trees	(128)		(128)	(128)	(122)	(122)	(122)	(122)	(122)
Understory	(3)		(3)	(3)	(4)	(4)	(4)	(4)	(4)
Forest Floor	(7)		(7)	(7)	8	8	8	8	8
Down Dead Wood	(15)		(15)	(15)	(16)	(16)	(16)	(16)	(16)
Forest Soils	(58)		(58)	(58)	(15)	(15)	(15)	(15)	(15)
Harvested Wood	**(57)**		**(56)**	**(56)**	**(58)**	**(56)**	**(59)**	**(57)**	**(58)**
Wood Products	(13)		(15)	(15)	(16)	(14)	(17)	(16)	(16)
Landfilled Wood	(44)		(41)	(41)	(42)	(42)	(42)	(41)	(42)
Total Net Flux	**(268)**		**(267)**	**(267)**	**(207)**	**(205)**	**(208)**	**(206)**	**(207)**

Note: 1 Tg C = 1 Tg carbon = 1 million metric tons carbon. This table has been included to facilitate comparison with previous U.S. Inventories. Parentheses indicate net carbon "sequestration" (i.e., accumulation into the carbon pool minus emissions or harvest from the carbon pool). The sum of the net stock changes in this table (i.e., total net flux) is an estimate of the actual net flux between the total forest carbon pool and the atmosphere. Shaded areas indicate values based on a combination of historical data and projections. Forest values are based on periodic measurements; harvested wood estimates are based on annual surveys. Totals may not sum due to independent rounding.

Table 6-5 presents the carbon stock estimates for forest and harvested wood storage pools. Together, the tree and forest soil pools account for over 80 percent of total carbon stocks. Carbon stocks in all pools, except forest floor, increased over time, indicating that during these periods, all storage pools, except forest floor, accumulated carbon (e.g., carbon sequestration by trees was greater than carbon removed from the tree pool through respiration, decay, litterfall, and harvest). Figure 6-2 shows 1997 forest carbon stocks, excluding harvested wood stocks, by the regions that were used in the forest carbon analysis. Figure 6-3 shows 1997 forest carbon stocks per hectare, by county, excluding harvested wood stocks, for all counties in the conterminous United States that have at least 5 percent of the county area in forest.

Table 6-5: U.S. Forest Carbon Stock Estimates (Tg C)

Carbon Pool	1987	1997	2002
Forest	**47,595**	**49,695**	**50440**

Trees	15,168	16,449	17059
Understory	448	473	493
Forest Floor	4,240	4,306	4,266
Down dead wood	2,058	2,205	2,285
Forest Soils	25,681	26,262	26,337
Harvested Wood	**1,920**	**2,478**	**2,767**
Wood Products	1,185	1,319	1,398
Landfilled Wood	735	1,159	1,369
Total Forest Carbon Stocks	**49,515**	**52,173**	**53,207**

Note: Forest carbon stocks do not include forest stocks in Alaska, Hawaii, or U.S. territories, or trees on non-forest land (e.g., urban trees); wood product stocks include exports, even if the logs are processed in other countries, and exclude imports. Shaded areas indicate values based on a combination of historical data and projections. All other estimates are based on historical data only. Totals may not sum due to independent rounding. Note that the stock is listed for 2002 because stocks are defined as of January 1 of the listed year.

Figure 6-2. Forest Carbon Stocks by Region, 1997

Figure 6-3. Forest Carbon Stocks, Per Hectare, by County, 1997

Methodology

The methodology described herein is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). The IPCC identifies two approaches for developing estimates of net carbon flux or stock change from Land-Use Change and Forestry: 1) using average annual statistics on land use, land-use change, and forest management activities, and applying carbon density and flux rate data to these activity estimates to derive total flux values; or 2) using carbon stock estimates derived from periodic inventories of forest stocks, and measuring net changes in carbon stocks over time. The latter approach was employed because the United States conducts periodic surveys of national forest stocks. In addition, the IPCC identifies two approaches to accounting for carbon emissions from harvested wood: 1) assuming that all of the harvested wood replaces wood products that decay in the inventory year so that the amount of carbon in annual harvests equals annual emissions from harvests; or 2) accounting for the variable rate of decay of harvested wood according to its disposition (e.g., product pool, landfill, combustion). The latter approach was applied for this Inventory using estimates of carbon stored in wood products and landfilled wood.[4] The use of direct measurements from forest surveys to estimate the forest pools, and the use of data on wood products and landfilled wood to estimate the harvested wood pool is likely to result in more accurate flux estimates than the alternative IPCC methodologies. Due to differences in data sources, different methods were used to calculate the carbon flux in forests and in harvested wood products. Therefore these methods are described separately below.

Forest Carbon Stock Change

The overall approach was to sample the forest carbon at one time, sample the forest carbon again several years later, and then subtract the two estimates to calculate the net change in carbon stocks. Three periodic inventories

[4] The product estimates in this study do not account for carbon stored in imported wood products. However, they do include carbon stored in exports, even if the logs are processed in other countries (Heath et al. 1996).

(sampling times) were used: 1987, 1997, and 2002.[5] For each periodic inventory, each carbon pool was estimated using coefficients from the FORCARB model, as described below. The carbon pools included live and dead standing trees, understory vegetation, forest floor, and soil. These estimates were summed to calculate total carbon stocks at each time period. Data sources and methods for estimating each carbon pool are described briefly below and more fully in Annex O.

The starting point for estimating forest carbon stock change was to obtain data on the area and growing stock volume for forest lands. For 1987 and 1997, such data were available from periodic inventories conducted by the USDA Forest Service Forest Inventory and Analysis program (Smith et al. 2001, Frayer and Furnival 1999). In the past, the Forest Inventory and Analysis program did not conduct detailed surveys of all forest land, but instead focused on timber producing land, which is called timberland. In addition, some reserved forest land and some other forest land were surveyed.[6] With the introduction of the new annualized inventory design (Gillespie 1999), all forest lands will feature the same type of detailed information. In order to include all forest lands, estimates were made for timberlands and then were extrapolated for non-timberland forests.

The Forest Inventory and Analysis program has conducted consistent forest surveys based on extensive statistically based sampling of much of the forest land in the United States since 1952. Historically, these were conducted periodically, state-by-state within a region. One state within a region would be surveyed, and when finished, another state was surveyed. Eventually (every 5 to 14 years, depending on the state), all states within a region would be surveyed, and then states would be resurveyed. The Forest Inventory and Analysis program has adopted a new annualized design, so that a portion of each state will be surveyed each year (Gillespie 1999); however, data are not yet available for all states. The annualized survey also includes a plan to measure attributes that are needed to estimate carbon in various pools, such as soil carbon and forest floor carbon. Currently, some of these pools must be estimated based on other measured characteristics. Characteristics that were measured in the 1987 and 1997 surveys include individual tree diameter and species, and forest type and age of the plot. For more information about forest inventory data and carbon stock change, see Birdsey and Heath (2001).

Historically, the main purpose of the Forest Inventory and Analysis program has been to estimate areas, volume of growing stock, and timber products output and utilization factors. Growing stock is a classification of timber inventory that includes live trees of commercial species meeting specified standards of quality (Smith et al. 2001). Timber products output refers to the production of industrial roundwood products such as logs and other round timber generated from harvesting trees, and the production of bark and other residue at processing mills. Utilization factors relate inventory volume to the volume cut or destroyed when producing roundwood (May 1998). Growth, harvests, land-use change, and other estimates of change are derived from repeated surveys.

For the 2002 periodic inventory, data were not available from the Forest Inventory and Analysis program. Therefore, areas, volumes, growth, land-use changes, and other forest characteristics were projected with a system of models representing the U.S. forest sector (see Haynes 2002, also see Annex O).

Based on the measured or projected periodic survey data, estimates were made of the total biomass and carbon in trees on timberlands and other forest lands. For timberlands, total biomass and carbon in standing trees were

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[5] As explained in the paragraphs below, the 1987 and 1997 "inventories" referred to here are actual forest inventories (i.e., datasets based on field surveys), while the 2002 "inventory" is a projection derived from the historical field data and a linked system of forest sector models. A national (field-based) forest inventory has not been completed for 2002.

[6] Forest land in the United States includes all land that is at least 10 percent stocked with trees of any size. Timberland is the most productive type of forest land, growing at a rate of 20 cubic feet per acre per year or more. In 1997, there were about 503 million acres of timberlands, which represented 67 percent of all forest lands (Smith and Sheffield 2000). Forest land classified as timberland is unreserved forest land that is producing or is capable of producing crops of industrial wood. The remaining 33 percent of forest land is classified as reserved forest land, which is forest land withdrawn from timber use by statute or regulation, or other forest land, which includes forests on which timber is growing at a rate less than 20 cubic feet per acre per year.

calculated from the growing stock volume. Calculations were made using biomass conversion factors for each forest type and region presented in Smith et al. (in press). For non-timberlands, biomass and carbon in standing trees were estimated based on average carbon estimates derived from similar timberlands. Reserved forests were assumed to contain the same average carbon densities as timberlands of the same forest type, region, and owner group. These averages were multiplied by the areas of non-timberland forests and then aggregated for a national total. Average carbon stocks were derived for other forest land by using average carbon stocks for Timberlands, which were multiplied by 50 percent to simulate the effects of lower productivity.

Understory carbon was estimated from inventory data using equations presented in Birdsey (1992). Forest floor carbon was estimated from inventory data using the equations presented in Smith and Heath (2002). Down dead wood was estimated using a procedure similar to that used for estimating carbon in understory vegetation, as described in Annex O. Data on the carbon content of soils were obtained from the national STATSGO spatial database. These data were combined with spatial data from the Forest Inventory and Analysis program on the location of U.S. forest lands to estimate soil carbon in all forest lands.

Once carbon pools were estimated as described above for each periodic inventory (1987, 1997, and 2002), the pools were summed together to create total forest carbon stock estimates. Average annual carbon stock changes were then calculated by subtracting carbon stocks at the end of a time period from those at the beginning of the time period, and then dividing by the number of years in the time period.

Harvested Wood Products Carbon Stock Change

Estimates of carbon stock changes in wood products and wood discarded in landfills were based on the methods described in Skog and Nicholson (1998). These methods utilize two harvested wood carbon storage pools: wood products in use, and wood discarded in landfills. Annual historical estimates and projections of detailed product production were used to divide consumed roundwood into product, wood mill residue, and pulp mill residue. Rates of decay for wood products and for wood in landfills were estimated and applied to the respective pools. The results were aggregated to produce national estimates. To account for imports and exports, the production approach was used, meaning that carbon in exported wood was included using the same disposal rates as in the United States, while carbon in imported wood was not included. Over the period 1990 through 2001, carbon in exported wood accounted for an average of 22 Tg CO_2 Eq. storage per year, with little variation from year to year. For comparison, imports (which were not included in the harvested wood net flux estimates) increased from 26 Tg CO_2 Eq. per year in 1990 to 47 Tg CO_2 Eq. per year in 2001.

Data Sources

The estimates of forest carbon stocks used in this Inventory to calculate forest carbon fluxes are based largely on areas, volumes, growth, harvests, and utilization factors derived from the forest inventory data collected by the USDA Forest Service Forest Inventory and Analysis program. Compilations of these data for 1987 and 1997 are given in Waddell et al. (1989) and Smith et al. (2001), with trends discussed in the latter citation. The timber volume data used here include timber volumes on forest land classified as timberland, as well as on some reserved forest land and other forest land. Timber volumes on forest land in Alaska, Hawaii, and the U.S. territories are not sufficiently detailed to be used here. Also, timber volumes on non-forest land (e.g., urban trees, rangeland) are not included. The timber volume data include estimates by tree species, size class, and other categories. The forest inventory data are used to derive estimates of carbon stocks as described above in the methodology section and in Annex O. Estimates of soil carbon are based on data from the STATSGO database (USDA 1991). Carbon stocks in wood products in use and wood products stored in landfills are based on historical data from the USDA Forest Service (USDA 1964, Ulrich 1989, Howard 2001), and historical data as implemented in the framework underlying the NAPAP (Ince 1994) and TAMM/ATLAS (Haynes 2002, Mills and Kincaid 1992) models. The carbon conversion factors and decay rates for harvested carbon removed from the forest are taken from Skog and Nicholson (1998).

Uncertainty

This section discusses uncertainties in the carbon sequestration estimates, given the methods and data used. There are sampling and measurement errors associated with the forest survey data that underlie the forest carbon estimates. These surveys are based on a statistical sample designed to represent the wide variety of growth conditions present over large territories. Although newer inventories are being conducted annually in every state, many of the data currently used were collected over more than one year in a state, and data associated with a particular year may actually have been collected over several previous years. Thus, there is uncertainty in the year associated with the forest inventory data. In addition, the forest survey data that are currently available generally exclude timber stocks on most forest land in Alaska, Hawaii, and U.S. territories. However, net carbon fluxes from these stocks are believed to be minor. The assumptions that were used to calculate carbon stocks in reserved forests and other forests in the conterminous United States also contribute to the uncertainty. Although the potential for uncertainty is large, the sample design for the forest surveys contributes to limiting the error in carbon flux. Estimates from sampling at different times on permanent plots are correlated, and such correlation reduces the uncertainty in estimates of carbon flux. For example, in a study on the uncertainty of the forest carbon budget of private Timberlands of the United States, Smith and Heath (2000) estimated that the uncertainty of the flux decreased more than three-fold when the correlation coefficient increased from 0.5 to 0.95.

Additional sources of uncertainty come from the models used to estimate carbon storage in specific ecosystem components, such as forest floor, understory vegetation, and soil. Extrapolation of the results of site-specific ecosystem studies to all forest lands introduces uncertainty because such studies may not adequately represent regional or national averages. Uncertainty also arises due to (1) modeling errors, for example relying on coefficients or relationships that are not well known, and (2) errors in converting estimates from one reporting unit to another (Birdsey and Heath 1995). An important source of uncertainty is that the impacts of forest management activities, including harvest, on soil carbon are not well understood. For example, while Johnson and Curtis (2001) found little or no net change in soil carbon following harvest on average across a number of studies, many of the individual studies did exhibit differences. Heath and Smith (2000b) noted that the experimental design in a number of soil studies was such that the usefulness of the studies may be limited in determining harvesting effects on soil carbon. Soil carbon impact estimates need to be very precise because even small changes in soil carbon may sum to large differences over large areas. This analysis assumes that soil carbon density for each forest type stays constant over time. As more information becomes available, the effects of changes in land use will be better accounted for in estimates of soil carbon.

Recent studies have begun to quantify the uncertainty in national-level carbon budgets based on the methods adopted here. Smith and Heath (2000) and Heath and Smith (2000a) report on an uncertainty analysis they conducted on carbon sequestration in private timberlands. These studies are not strictly comparable to the estimates in this chapter because they used an older version of the FORCARB model, which was based on older data and produced decadal estimates. However, the magnitudes of the uncertainties should be instructive. Their results indicate that the carbon flux of private timberlands, not including harvested wood, was approximately the average carbon flux (271 Tg CO_2 Eq. per year) ±15 percent at the 80 percent confidence level for the period 1990 through 1999. The flux estimate included the tree, soil, understory vegetation, and forest floor components only. The uncertainty in the carbon inventory of private timberlands for 2000 was approximately 5 percent at the 80 percent confidence level. These estimates did not include all uncertainties, such as the ones associated with public timberlands, and reserved and other forest land, but they did include many of the types of uncertainties listed previously. Because of these additional factors, uncertainty is expected to be greater in estimates for all forest lands.

Changes in Carbon Stocks in Urban Trees

Urban forests constitute a significant portion of the total U.S. tree canopy cover (Dwyer et al. 2000). It is estimated that urban areas (cities, towns, and villages), which cover 3.5 percent of the continental United States, contain about 3.8 billion trees. With an average tree canopy cover of 27.1 percent, urban areas account for approximately 3 percent of total tree cover in the continental United States.

Trees in urban areas of the continental United States were estimated by Nowak and Crane (2001) to account for an average annual net sequestration of 59 Tg CO_2 Eq. (16 Tg C). This estimate is representative of the period from 1990 through 2001, as it is based on data collected during the 1990s. Annual estimates of CO_2 flux have not been developed (Table 6-6).

Table 6-6: Net CO_2 Flux From Urban Trees (Tg CO_2 Eq.)

Year	Tg CO_2 Eq.
1990	(58.7)
1995	(58.7)
1996	(58.7)
1997	(58.7)
1998	(58.7)
1999	(58.7)
2000	(58.7)
2001	(58.7)

Note: Parentheses indicate net sequestration.

Methodology

The methodology used by Nowak and Crane (2001) is based on average annual estimates of urban tree growth and decomposition, which were derived from field measurements and data from the scientific literature, urban area estimates from U.S. Census data, and urban tree cover estimates from remote sensing data. This approach is consistent with, but more robust than, the default IPCC methodology in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).[7]

Nowak and Crane (2001) developed estimates of annual gross carbon sequestration from tree growth and annual gross carbon emissions from decomposition for ten U.S. cities: Atlanta, GA; Baltimore, MD; Boston, MA; Chicago, IL; Jersey City, NJ; New York, NY; Oakland, CA; Philadelphia, PA, Sacramento, CA; and Syracuse, NY. The gross carbon sequestration estimates were derived from field data that were collected in these ten cities during the period from 1989 through 1999, including tree measurements of stem diameter, tree height, crown height, and crown width, and information on location, species, and canopy condition. The field data were converted to annual gross carbon sequestration rates for each species (or genus), diameter class, and land-use condition (forested, park-like, and open growth) by applying allometric equations, a root-to-shoot ratio, moisture contents, a carbon content of 50 percent (dry weight basis), an adjustment factor to account for smaller aboveground biomass volumes (given a particular diameter) in urban conditions compared to forests, an adjustment factor to account for tree condition (fair to excellent, poor, critical, dying, or dead), and annual diameter and height growth rates. The annual gross carbon sequestration rates for each species (or genus), diameter class, and land-use condition were then scaled up to city estimates using tree population information.

The annual gross carbon emission estimates were derived by applying to carbon stock estimates, which were derived as an intermediate step in the gross sequestration calculations, estimates of annual mortality by tree diameter and condition class, assumptions about whether dead trees would be removed from the site—since removed trees were assumed to decay faster than those left on the site—and assumed decomposition rates for dead trees left standing and dead trees that are removed. The annual gross carbon emission rates for each species (or genus), diameter class, and condition class were then scaled up to city estimates using tree population information.

[7] It is more robust in that both growth and decomposition are accounted for, and data from individual trees are scaled up to state and then national estimates based on data on urban area and urban tree canopy cover.

Annual net carbon sequestration estimates were derived for seven of the ten cities by subtracting the annual gross emission estimates from the annual gross sequestration estimates.[8]

National annual net carbon sequestration by urban trees was estimated from the city estimates of gross and net sequestration, and urban area and urban tree cover data for the contiguous United States. Note that the urban areas are based on U.S. Census data, which define "urban" as having a population density greater than 1,000 people per square mile or population total greater than 2,500. Therefore, urban encompasses most cities, towns, and villages (i.e., it includes both urban and suburban areas). The gross and net carbon sequestration values for each city were divided by each city's area of tree cover to determine the average annual sequestration rates per unit of tree area for each city. The median value for gross sequestration (0.30 kg C/m^2-year) was then multiplied by an estimate of national urban tree cover area (76,151 km^2) to estimate national annual gross sequestration. To estimate national annual net sequestration, the estimate of national annual gross sequestration was multiplied by the average of the ratios of net to gross sequestration for those cities that had both estimates. The average of these ratios is 0.70.

Table 6-7: Carbon Stocks (Metric Tons C), Annual Carbon Sequestration (Metric Tons C/yr), Tree Cover (Percent), and Annual Carbon Sequestration per Area of Tree Cover (kg C/m^2 cover-yr) for Ten U.S. Cities

City	Carbon Stocks	Gross Annual Sequestration	Net Annual Sequestration	Tree cover	Gross Annual Sequestration per Area of Tree Cover	Net Annual Sequestration per Area of Tree Cover
New York, NY	1,225,200	38,400	20,800	20.9	0.23	0.12
Atlanta, GA	1,220,200	42,100	32,200	36.7	0.34	0.26
Sacramento, CA	1,107,300	20,200	NA	13.0	0.66	NA
Chicago, IL	854,800	40,100	NA	11.0	0.61	NA
Baltimore, MD	528,700	14,800	10,800	25.2	0.28	0.20
Philadelphia, PA	481,000	14,600	10,700	15.7	0.27	0.20
Boston, MA	289,800	9,500	6,900	22.3	0.30	0.22
Syracuse, NY	148,300	4,700	3,500	24.4	0.30	0.22
Oakland, CA	145,800	NA	NA	21.0	NA	NA
Jersey City, NJ	19,300	800	600	11.5	0.18	0.13

NA = not analyzed

Data Sources

The field data from the 10 cities, some of which are unpublished, are described in Nowak and Crane (2001) and references cited therein. The allometric equations were taken from the scientific literature (see Nowak 1994, Nowak et al. 2002), and the adjustments to account for smaller volumes in urban conditions were based on information in Nowak (1994). A root-to-shoot ratio of 0.26 was taken from Cairns et al. (1997), and species- or genus-specific moisture contents were taken from various literature sources (see Nowak 1994). Adjustment factors to account for tree condition were based on percent crown dieback (Nowak and Crane 2001). Tree growth rates were also taken from existing literature. Average diameter growth was based on the following sources: estimates for trees in forest stands came from Smith and Shifley (1984); estimates for trees on land uses with a park-like structure came from deVries (1987); and estimates for more open-grown trees came from Nowak (1994). Formulas from Fleming (1988) formed the basis for average height growth calculations. Estimates of annual mortality rates by diameter class and condition class were derived from a study of street-tree mortality (Nowak 1986). Assumptions about whether dead trees would be removed from the site were based on expert judgment of the authors. Decomposition rates were based on literature estimates (Nowak and Crane 2001). Urban tree cover area estimates for each of the 10 cities and the contiguous United States were obtained from Dwyer et al. (2000) and Nowak et al. (2001).

[8] Three cities did not have net estimates.

Uncertainty

The estimates are based on limited field data collected in ten U.S. cities, and the uncertainty in these estimates increases as they are scaled up to the national level. There is also uncertainty associated with the biomass equations, conversion factors, and decomposition assumptions used to calculate carbon sequestration and emission estimates (Nowak et al. 2002), as well as with the tree cover area estimates for urban areas, as these are based on interpretation of Advanced Very High Resolution Radiometer data. In addition, these results do not include changes in soil carbon stocks, and there may be some overlap between the urban tree carbon estimates and the forest tree carbon estimates. However, both the omission of urban soil carbon flux, and the potential overlap with forest carbon, are believed to be relatively minor (Nowak 2002).

Changes in Agricultural Soil Carbon Stocks

The amount of organic carbon contained in soils depends on the balance between inputs of organic material (e.g., decayed plant matter, roots, and organic amendments such as manure and crop residues) and loss of carbon through decomposition. The quantity and quality of organic matter inputs, and their rate of decomposition, are determined by the combined interaction of climate, soil properties, and land use. Agricultural practices such as clearing, drainage, tillage, planting, grazing, crop residue management, fertilization, and flooding, can modify both organic matter inputs and decomposition, and thereby result in a net flux of carbon to or from soils. In addition, the application of carbonate minerals to soils through liming operations results in emissions of CO_2. The IPCC methodology for estimation of net CO_2 flux from agricultural soils (IPCC/UNEP/OECD/IEA 1997) is divided into three categories of land-use/land-management activities: 1) agricultural land-use and land-management activities on mineral soils; 2) agricultural land-use and land-management activities on organic soils; and 3) liming of soils. Mineral soils and organic soils are treated separately because each responds differently to land-use practices.

Mineral soils contain comparatively low amounts of organic matter (usually less than 20 percent by weight), much of which is concentrated near the soil surface. Typical well-drained mineral surface soils contain from 1 to 6 percent organic matter (by weight); mineral subsoils contain even lower amounts of organic matter (Brady and Weil 1999). When mineral soils undergo conversion from their native state to agricultural use, as much as half of the soil organic carbon can be lost to the atmosphere. The rate and ultimate magnitude of carbon loss will depend on native vegetation, conversion method and subsequent management practices, climate, and soil type. In the tropics, 40 to 60 percent of the carbon loss generally occurs within the first 10 years following conversion; after that, carbon stocks continue to decline but at a much slower rate. In temperate regions, carbon loss can continue for several decades. Eventually, the soil will reach a new equilibrium that reflects a balance between carbon accumulation from plant biomass and carbon loss through oxidation. Any changes in land-use or management practices that result in increased organic inputs or decreased oxidation of organic matter (e.g., 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.

Organic soils, which are also referred to as histosols, include all soils with more than 20 to 30 percent organic matter by weight, depending on clay content (Brady and Weil 1999). The organic matter layer of these soils is also typically extremely deep. Organic soils form under water-logged conditions, in which decomposition of plant residues is retarded. When organic soils are cultivated, they are first drained which, together with tilling or mixing of the soil, aerates the soil, and thereby accelerates the rate of decomposition and CO_2 generation. Because of the depth and richness of the organic layers, carbon loss from cultivated organic soils can continue over long periods of time. When organic soils are disturbed, through cultivation and/or drainage, the rate at which organic matter decomposes, and therefore the rate at which CO_2 emissions are generated, is determined primarily by climate, the composition (i.e., decomposability) of the organic matter, and the specific land-use practices undertaken. The use of organic soils for annual crops results in greater carbon loss than conversion to pasture or forests, due to deeper drainage and more intensive management practices (Armentano and Verhoeven 1990, as cited in IPCC/UNEP/OECD/IEA 1997).

Lime in the form of crushed limestone ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$) is commonly added to agricultural soils to ameliorate acidification. When these compounds come in contact with acid soils, they degrade, thereby generating CO_2. The rate of degradation is determined by soil conditions and the type of mineral applied; it can take several years for applied limestone and dolomite to degrade completely.

Of the three activities, use and management of mineral soils was the most important component of total flux during the 1990 through 2001 period. Carbon sequestration in mineral soils in 2001 was estimated at approximately 59 Tg CO_2 Eq. (16 Tg C), while emissions from organic soils were estimated at 35 Tg CO_2 Eq. (9 Tg C) and emissions from liming were estimated at 9 Tg CO_2 Eq. (2.5 Tg C). Together, the three activities accounted for net sequestration of approximately 15 Tg CO_2 Eq. (4 Tg C) in 2001. Total annual net CO_2 flux was negative (i.e., net sequestration) each year over the 1990 to 2001 period. Between 1990 and 2001, total net carbon sequestration in agricultural soils increased by close to 14 percent. The increase is largely due to additional acreage of annual cropland converted to permanent pastures and hay production, a reduction in the frequency of summer-fallow use in semi-arid areas and some increase in the adoption of conservation tillage (i.e., reduced and no-till) practices. The relatively large shift in annual net sequestration from 1990 to 1995 is the result of calculating average annual mineral and organic soil fluxes from periodic, rather than annual, activity data.[9]

The spatial variability in annual, per hectare CO_2 flux for mineral and organic soils is displayed in Figure 6-4 through Figure 6-7. The greatest mineral soil sequestration rates are in the south and east central United States and in a small area of the Pacific Northwest, while the greatest organic soil emission rates are along the southeast coast, in the northeast central United States, and along the central west coast.

Table 6-8: Net CO_2 Flux From Agricultural Soils (Tg CO_2 Eq.)

	1990		1995	1996	1997	1998	1999	2000	2001
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 areas indicate values based on a combination of historical data and projections. All other values are based on historical data only.

Figure 6-4. Net Annual CO_2 Flux, per Hectare, From Mineral Soils Under Agricultural Management, 1990-1992

Figure 6-5. Net Annual CO_2 Flux, per Hectare, From Mineral Soils Under Agricultural Management, 1993-2001

Figure 6-6. Net Annual CO_2 Flux, per Hectare, From Organic Soils Under Agricultural Management, 1990-1992

Figure 6-7. Net Annual CO_2 Flux, per Hectare, From Organic Soils Under Agricultural Management, 1993-2001

The flux estimates presented here are restricted to CO_2 fluxes associated with the use and management of agricultural soils. Agricultural soils are also important sources of other greenhouse gases, particularly nitrous oxide (N_2O) from application of fertilizers, manure, and crop residues and from cultivation of legumes, as well as methane (CH_4) from flooded rice cultivation. These emissions are accounted for in the Agriculture chapter.[10] It should be noted that other land-use and land-use change activities result in fluxes of non-CO_2 greenhouse gases to and from soils that are not currently accounted for. These include emissions of CH_4 and N_2O from managed forest soils (above what would occur if the forest soils were undisturbed), as well as CH_4 emissions from artificially flooded lands, resulting from activities such as dam construction. Aerobic (i.e., non-flooded) soils are a sink for CH_4, so soil

[9] Mineral and organic soil results for the entire time series are presented in Annex P.

[10] Nitrous oxide emissions from agricultural soils and methane emissions from rice fields are addressed under the Agricultural Soil Management and Rice Cultivation sections, respectively, of the Agriculture chapter.

drainage can result in soils changing from a CH_4 source to a CH_4 sink, but if the drained soils are used for agriculture, fertilization and tillage, disturbance can reduce the ability of soils to oxidize CH_4. The non-CO_2 emissions and sinks from these other land use and land-use change activities were not assessed due to scientific uncertainties about the greenhouse gas fluxes that result from these activities.

Methodology and Data Sources

The methodologies used to calculate net CO_2 flux from use and management of mineral and organic soils and from liming follow the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997, Ogle et al. 2002, Ogle et al. in review), except where noted below. (Additional details on the methodology and data used to estimate flux from mineral and organic soils are described in Annex P). Mineral soil organic carbon stocks were estimated for 1982, 1992, and 1997 for the conterminous United States and Hawaii using U.S. data on climate, soil types, land use and land management activity data, reference carbon stocks (for agricultural soils rather than native soils) and field studies addressing management effects on soil organic carbon storage. National-scale data on land-use and management changes over time were obtained from the *1997 National Resources Inventory* (NRCS 2000). The *1997 National Resources Inventory* provides land use/management data and soils information for more than 400,000 locations in U.S. agricultural lands. Two other sources were used to supplement the land-use information from the *1997 National Resources Inventory*. The Conservation Technology Information Center (CTIC 1998) provided data on tillage activity, with adjustments for long-term adoption of no-till agriculture (Towery 2001), and Euliss and Gleason (2002) provided activity data on wetland restoration of Conservation Reserve Program Lands. Major Land Resource Areas (MLRAs, NRCS 1981) were used as the base spatial unit for mapping climate regions in the United States. Each Major Land Resource Area represents a geographic unit with relatively similar soils, climate, water resources, and land uses (NRCS 1981).[11] Major Land Resource Areas were classified into climate zones according to the IPCC categories using the Parameter-Evaluation Regressions on Independent Slopes Model (PRISM) climate-mapping program of Daly et al. (1994). Reference carbon stocks were estimated using the National Soil Survey Characterization Database (NRCS 1997), and the reference condition for the stock estimates was cultivated cropland, rather than native vegetation as used in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). Changing the reference condition was necessary because soil measurements under agricultural management are much more common and easily identified in the National Soil Survey Characterization Database (NRCS 1997). Management factors were derived from published literature to determine the impact of management practices on soil organic carbon storage, including changes in tillage, cropping rotations and intensification, as well as land-use change between cultivated and uncultivated conditions (Ogle et al. in review). Euliss and Gleason (2002) provided the data for computing the change in soil organic carbon storage resulting from restoration of Conservation Reserve Program Lands (Olness et al. in press, Euliss et al. in prep). Combining information from these data sources, carbon stocks were estimated 50,000 times for 1982, 1992, and 1997, using a Monte Carlo simulation approach and the probability density functions for U.S.-specific management factors, reference carbon stocks, and land-use activity data (Ogle et al. in review, Ogle et al. 2002). The annual carbon flux for 1990 through 1992 was estimated by calculating the annual change in stocks between 1982 and 1992; annual carbon flux for 1993 through 2001 was estimated by calculating the annual change in stocks between 1992 and 1997 (see Table 6-9).

Table 6-9: Net Annual CO_2 Flux from U.S. Agricultural Soils Based on Monte Carlo Simulation (Tg CO_2 Eq.)

Soil Type	1990-1992	1993-2001
Mineral Soils		
Estimate*	(35.8)	(35.4)
Uncertainties	(13.9) to (58.7)	(20.9) to (50.3)
Organic Soils		
Estimate	34.3	34.8
Uncertainties	23.1 to 48.4	23.5 to 49.1
Total		
Estimate	(1.5)	(0.7)

[11] The polygons displayed in Figure 6-4 through Figure 6-7 are the Major Land Resource Areas.

Uncertainties	24.2 to (27.2)	19.5 to (19.5)

Note: Parentheses indicate net sequestration.
The uncertainties are based on the Monte Carlo analysis. The range is a 95 percent confience interval, based on the simulated values at the 2.5 and 97.5 percentiles in the final distributeion of 50,000 estimates
* Does not include the change in carbon storage resulting from the annual application of manure and sewage sludge, or the change in Conservation Reserve Program enrollment after 1997.

Annual carbon emission estimates from organic soils used for agriculture between 1990-2001 were derived using *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), except that U.S.-specific carbon loss rates were used in the calculations rather than default IPCC rates (Ogle et al. 2002). Similar to mineral soils, the final estimates include a measure of uncertainty as determined from the Monte Carlo simulation. Data from published literature were used to derive probability density functions for carbon loss rates (Ogle et al. in review), which were used to compute emissions based on the 1992 and 1997 land areas in each climate/land-use category defined in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). The area estimates were derived from the same climate, soil, and land-use/management databases that were used for mineral soil calculations (Daly et al. 1994, USDA 2000). The annual flux estimated for 1992 was applied to 1990 through 1992, and the annual flux estimated for 1997 was applied to 1993 through 2001 (see Table 6-9).

Annual carbon flux estimates for mineral soils between 1990 and 2001 were adjusted to account for additional carbon sequestration from manure and sewage sludge applications, as well as gains or losses in carbon sequestration due to changes in Conservation Reserve Program enrollment after 1997. The amount of land receiving manure and sewage sludge was estimated from nitrogen application data from the Agricultural Soil Management section of the Agriculture chapter of this volume, and an assumed application rate derived from Kellogg et al. (2000). The total land area was subdivided between cropland and grazing land based on supplemental information collected by the USDA (ERS 2000, NASS 2002). Carbon storage rate was estimated at 0.10 metric tons C per hectare per year for cropland and 0.33 metric tons C per hectare per year for grazing land. To estimate the carbon impacts of changes in Conservation Reserve Program enrollment after 1997, the changes in Conservation Reserve Program acreage relative to 1997 were derived based on Barbarika (2002), and the mineral soil changes were multiplied by 0.5 metric tons C per hectare per year.

Carbon dioxide emissions from degradation of limestone and dolomite applied to agricultural soils were calculated by multiplying the annual amounts of limestone and dolomite applied (see Table 6-10) by CO_2 emission factors (0.120 metric ton C/metric ton limestone, 0.130 metric ton C/metric ton dolomite).[12] These emission factors are based on the assumption that all of the carbon in these materials evolves as CO_2 in the same year in which the minerals are applied. The annual application rates of limestone and dolomite were derived from estimates and industry statistics provided in the *Minerals Yearbook* and *Mineral Industry Surveys* (Tepordei 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2002; USGS 2002). To develop these data, USGS (U.S. Bureau of Mines prior to 1997) obtained production and use information by surveying crushed stone manufacturers. Because some manufacturers were reluctant to provide information, the estimates of total crushed limestone and dolomite production and use were divided into three components: 1) production by end-use, as reported by manufacturers (i.e., "specified" production); 2) production reported by manufacturers without end-uses specified (i.e., "unspecified" production); and 3) estimated additional production by manufacturers who did not respond to the survey (i.e., "estimated" production).

To estimate the "unspecified" and "estimated" amounts of crushed limestone and dolomite applied to agricultural soils, it was assumed that the fractions of "unspecified" and "estimated" production that were applied to agricultural soils in a specific year were equal to the fraction of "specified" production that was applied to agricultural soils in that same year. In addition, data were not available for 1990, 1992, and 2001 on the fractions of total crushed stone

[12] The default emission factor for dolomite provided in the Workbook volume of the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) is incorrect. The value provided is 0.122 metric ton carbon/metric ton of dolomite; the correct value is 0.130 metric ton carbon/metric ton of dolomite.

production that were limestone and dolomite, and on the fractions of limestone and dolomite production that were applied to soils. To estimate the 1990 and 1992 data, a set of average fractions were calculated using the 1991 and 1993 data. These average fractions were applied to the quantity of "total crushed stone produced or used" reported for 1990 and 1992 in the 1994 *Minerals Yearbook* (Tepordei 1996). To estimate 2001 data, the 2000 fractions were applied to a 2001 estimate of total crushed stone presented in the USGS *Mineral Industry Surveys: Crushed Stone and Sand and Gravel in the First Quarter of 2002* (USGS 2002).

The primary source for limestone and dolomite activity data is the *Minerals Yearbook*, published by the Bureau of Mines through 1994 and by the U.S. Geological Survey from 1995 to the present. In 1994, the "Crushed Stone" chapter in *Minerals Yearbook* began rounding (to the nearest thousand) quantities for total crushed stone produced or used. It then reported revised (rounded) quantities for each of the years from 1990 to 1993. In order to minimize the inconsistencies in the activity data, these revised production numbers have been used in all of the subsequent calculations.

Table 6-10: Quantities of Applied Minerals (Thousand Metric Tons)

Mineral	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**
Limestone	19,012	20,312	17,984	15,609	16,686	17,297	17,479	16,539	14,882	16,894	15,863	16,473
Dolomite	2,360	2,618	2,232	1,740	2,264	2,769	2,499	2,989	6,389	3,420	3,812	3,959

Uncertainty

Although the mineral and organic soil estimates have been improved from previous years using a Monte Carlo approach with the incorporation of U.S.-specific reference carbon stocks and management factor values, some limitations do remain in the analysis. First, minimal data exist on where and how much manure and sewage sludge has been applied to U.S. agricultural lands. Consequently, uncertainties have not been estimated for the change in soil organic carbon storage resulting from these applications. Second, due to the IPCC requirement that inventories include all land areas that are potentially subject to land-use change, the *1997 National Resources Inventory* dataset includes some points designated as non-agricultural land-uses if this designation changed during the period from 1992 to 1997. The non-agricultural land uses are urban, water, and miscellaneous non-cropland (e.g., roads and barren areas). The impact on carbon storage resulting from converting cropland to non-agricultural uses is not well understood, and therefore, those points were not included in the calculations. Third, this inventory may underestimate losses of carbon from organic soils because the *1997 National Resources Inventory* was not designed as a soil survey and organic soils frequently occur as relatively small inclusions within major soil types. Lastly, this methodology does not take into account changes in carbon stocks due to pre-1982 land use and land-use change.

A revised inventory approach to better quantify uncertainty and to better represent between-year variability in annual fluxes is being developed and is currently under review. This new annual activity-based inventory, using a dynamic simulation model, would use climate, soil, and land-use/land-management databases to estimate annual variation in fluxes and include the effects of long-term trends in agricultural productivity on soil carbon stocks (see Box 6-1).

Uncertainties in the estimates of emissions from liming result from both the methodology and the activity data. The IPCC method assumes that all the inorganic carbon in the applied minerals evolves to CO_2, and that this degradation occurs in the same year that the minerals are applied. However, recent research has shown that liming can either be a carbon source or a sink, depending upon weathering reactions, which are pH dependent (Hamilton et al. 2002). Moreover, it can take several years for agriculturally applied limestone and dolomite to degrade completely. However, application rates are fairly constant over the entire time series, so this latter assumption may not contribute significantly to overall uncertainty.

There are several sources of uncertainty in the limestone and dolomite activity data. When reporting data to the USGS (or U.S. Bureau of Mines), some producers do not distinguish between limestone and dolomite. In these cases, data are reported as limestone, so this could lead to an overestimation of limestone and an underestimation of dolomite. In addition, the total quantity of crushed stone listed each year in the *Minerals Yearbook* excludes

American Samoa, Guam, Puerto Rico, and the U.S. Virgin Islands. The *Mineral Industry Surveys* further excludes Alaska and Hawaii from its totals.

[BEGIN BOX]

Box 6-1: Century model estimates of soil carbon stock changes on cropland

Soil carbon stock changes on U.S. cropland were estimated using a dynamic ecosystem simulation model called Century (Metherell et al. 1993, Parton et al. 1994). This method differs from the IPCC approach in that annual changes are computed dynamically as a function of inputs of carbon to soil (e.g., crop residues, manure) and carbon emissions from organic matter decomposition, which are governed by climate and soil factors as well as management practices. The model simulates all major field crops (maize, wheat and other small grains, soybean, sorghum, cotton) as well as hay and pasture (grass, alfalfa, clover). Management variables included tillage, fertilization, irrigation, drainage, and manure addition.

Input data were largely from the same sources as in the IPCC-based method (i.e., climate variables were from the PRISM database; crop rotation, irrigation and soil characteristics were from the National Resources Inventory (NRI); and tillage data were from the Conservation Technology Information Center (CTIC). In addition, the Century analysis used detailed information on crop rotation-specific fertilization and tillage implements obtained from USDA's Economic Research Service. The main difference between the methods is that the climate, soil and management data serve as 'driving variables' in the Century simulation, whereas in the IPCC approach these data are more highly aggregated and are used for classification purposes. In the Century-based analysis, land areas having less than 5 percent of total area in crop production were excluded and several less-dominant crops (e.g., vegetables, sugar beets and sugar cane, potatoes, tobacco, orchards, and vineyards), for which the model has not yet been parameterized, were not included. Thus, the total area included in the Century analysis (149 million hectares) was smaller than the corresponding area of cropland (165 million hectares) included in the IPCC estimates.

Preliminary results using the Century model suggest (as with the IPCC model) that U.S. cropland mineral soils (excluding organic soils) are currently acting as a carbon sink. The Century model estimates are that U.S. cropland soils sequester approximately 77 Tg CO_2 Eq. per year (21 Tg C/year) (average rates for 1992 through 1997). Organic soils (which contribute large C losses) were not simulated by Century.

As with the IPCC method, increases in mineral soil C stocks in the Century analysis are associated with reduced tillage, Conservation Reserve Program lands, reduced bare fallow and some increase in hay area. However, the Century analysis also includes the effect of a long-term trend in increasing residue inputs due to higher productivity on cropland in general, which contributes to the increase in soil carbon stocks. However, further work is needed to refine model input data and to estimate uncertainty for the dynamic model approach. Potential advantages of a dynamic simulation-based approach include the ability to use actual observed weather, observed annual crop yields, and more detailed soils and management information to drive the estimates of soil carbon change. This would facilitate annual estimates of carbon stock changes and CO_2 emissions from soils that would better reflect interannual variability in cropland production and weather influences on carbon cycle processes.

[END BOX]

Changes in Yard Trimming Carbon Stocks in Landfills

As is the case with carbon in landfilled forest products, carbon contained in landfilled yard trimmings can be stored indefinitely. In the United States, yard trimmings (i.e., grass clippings, leaves, branches) comprise a significant portion of the municipal waste stream, and a large fraction of the collected yard trimmings are discarded in landfills. However, both the amount of yard trimmings collected annually and the fraction that is landfilled have declined over the last decade. In 1990, nearly 32 million metric tons (wet weight) of yard trimmings were collected at landfills and transfer stations (Franklin Associates 1999). Since then, programs banning or discouraging disposal have led to an increase in backyard composting and the use of mulching mowers, and a consequent 21 percent decrease in the amount of yard trimmings collected. At the same time, a dramatic increase in the number of municipal composting facilities has reduced the proportion of collected yard trimmings that are discarded in landfills—from 72 percent in 1990 to 26 percent in 2001. The decrease in the yard trimmings landfill disposal rate has resulted in a decrease in the rate of landfill carbon storage from approximately 18 Tg CO_2 Eq. in 1990 to 5 Tg CO_2 Eq. in 2001 (Table 6-11).

Table 6-11: Net CO_2 Flux from Landfilled Yard Trimmings (Tg CO_2 Eq.)

Year	Tg CO_2 Eq.
1990	(18.2)
1995	(11.6)
1996	(9.7)
1997	(9.0)
1998	(8.7)
1999	(7.8)
2000	(6.9)
2001	(5.3)

Note: Parentheses indicate net storage. Shaded area indicates values based on projections.

Methodology

The methodology for estimating carbon storage is based on a life-cycle analysis of greenhouse gas emissions and sinks associated with solid waste management (EPA 1998). According to this methodology, carbon storage is the product of the weight of landfilled yard trimmings and a storage factor. The storage factor, which is the ratio of the weight of the carbon that is stored indefinitely to the wet weight of the landfilled yard trimmings, is based on a series of experiments designed to evaluate CH_4 generation and residual organic material in landfills (Barlaz 1998). These experiments analyzed grass, leaves, branches, and other materials, and were designed to promote biodegradation by providing ample moisture and nutrients.

Barlaz (1998) determined carbon storage factors, on a dry weight basis, for each of the three components of yard trimmings: grass, leaves, and branches (see Table 6-12). For purposes of this analysis, these were converted to wet weight basis using assumed moisture contents of 0.6, 0.2, and 0.4, respectively. To develop a weighted average carbon storage factor, the composition of yard trimmings was assumed to consist of 50 percent grass clippings, 25 percent leaves, and 25 percent branches on a wet weight basis. The weighted average carbon storage factor is 0.22 (weight of carbon stored indefinitely per unit weight of wet yard trimmings).

Table 6-12: Storage Factor (kg C/kg dry yard trimmings), Moisture Content (kg water/kg wet yard trimmings), Yard Trimmings Composition (percent), and Carbon Storage Factor (kg C/kg wet yard trimmings) of Landfilled Yard Trimmings

	Grass	Leaves	Branches
Storage Factor[a]	0.30	0.46	0.43
Moisture Content	0.60	0.20	0.40
Yard Trimmings Composition	50%	25%	25%
Converted Storage Factor[b]	0.12	0.37	0.26

[a] From Barlaz (1998), adjusted using CH_4 yields in Eleazer et al. (1997).

[b] The converted storage factor for each component is the product of the original storage factor and one minus the moisture content; the weighted average storage factor for yard trimmings is obtained by weighting the component storage factors by the yard trimmings composition percents.

Data Sources

The yard trimmings discards data were taken from two reports: *Characterization of Municipal Solid Waste in the United States: 1998 Update* (Franklin Associates 1999) and *Municipal Solid Waste in the United States: 2000 Facts and Figures* (EPA 2002), which provide estimates for 1990 through 2000 (see Table 6-13). Yard trimmings discards for 2001 were projected using a linear regression of the 1990 through 2000 data. These reports do not subdivide discards of individual materials into volumes landfilled and combusted, although they provide an estimate of the overall distribution of solid waste between these two management methods (i.e., ranging from 81 percent and 19 percent respectively in 1990, to 77 percent and 23 percent in 2001) for the waste stream as a whole.[13] Thus, yard trimmings disposal to landfills is the product of the quantity discarded and the proportion of discards managed in landfills. As discussed above, the carbon storage factor was derived from the results of Barlaz (1998) and Eleazer et al. (1977), and assumed moisture contents and component fractions for yard trimmings.

Table 6-13: Collection and Destination of Yard Trimmings (Million Metric Tons, or Tg, wet weight)

Destination	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
Municipal Composting Facilities	3.8	4.4	4.9	6.3	7.3	8.2	9.4	10.4	11.4	12.9	14.3	16.6
Discarded	27.9	27.4	26.9	23.9	21.3	18.8	15.9	14.7	13.8	12.3	10.9	8.6
Landfill	22.8	22.2	21.7	19.2	17.1	14.5	12.1	11.3	10.8	9.8	8.6	6.6
Incineration	5.2	5.2	5.2	4.7	4.2	4.3	3.8	3.4	2.9	2.5	2.3	1.9
Total	**31.8**	**31.8**	**31.8**	**30.2**	**28.6**	**26.9**	**25.3**	**25.2**	**25.2**	**25.2**	**25.2**	**25.2**

Note: Shaded area indicates values based on projections.

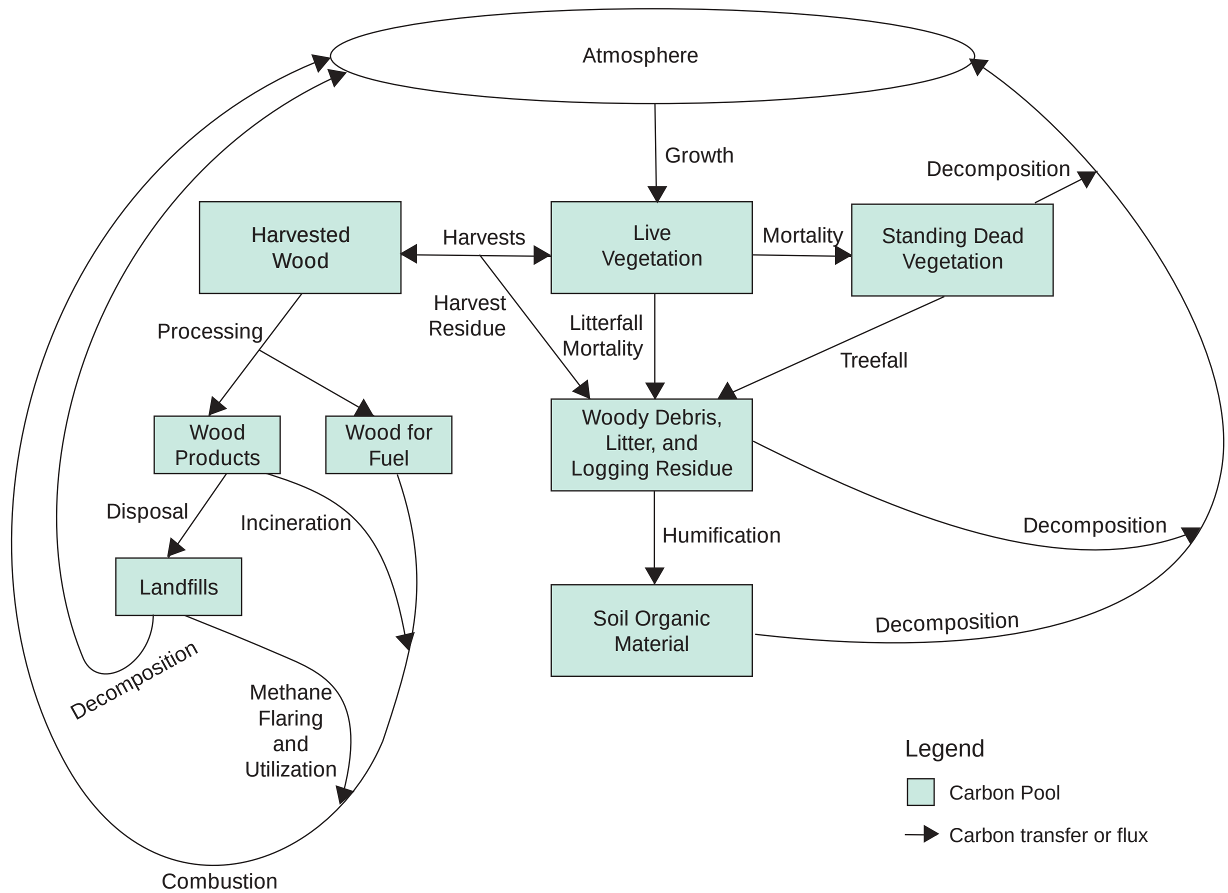
Uncertainty

The principal source of uncertainty for the landfill carbon storage estimates stems from an incomplete understanding of the long-term fate of carbon in landfill environments. Although there is ample field evidence that many landfilled organic materials remain virtually intact for long periods, the quantitative basis for predicting long-term storage is based on limited laboratory results under experimental conditions. In reality, there is likely to be considerable heterogeneity in storage rates, based on 1) actual composition of yard trimmings (e.g., oak leaves decompose more slowly than grass clippings) and 2) landfill characteristics (e.g., availability of moisture, nitrogen, phosphorus, etc.). Other sources of uncertainty include the estimates of yard trimmings disposal rates, which are based on extrapolations of waste composition surveys, and the extrapolation of values for 2001 disposal from estimates for the period from 1990 through 2000. In addition, the methodology does not include an accounting of changes in carbon stocks in yards.

[13] These percents represent the percent of total municipal solid waste (MSW) discards after recovery for recycling or composting.

Figure 6-1

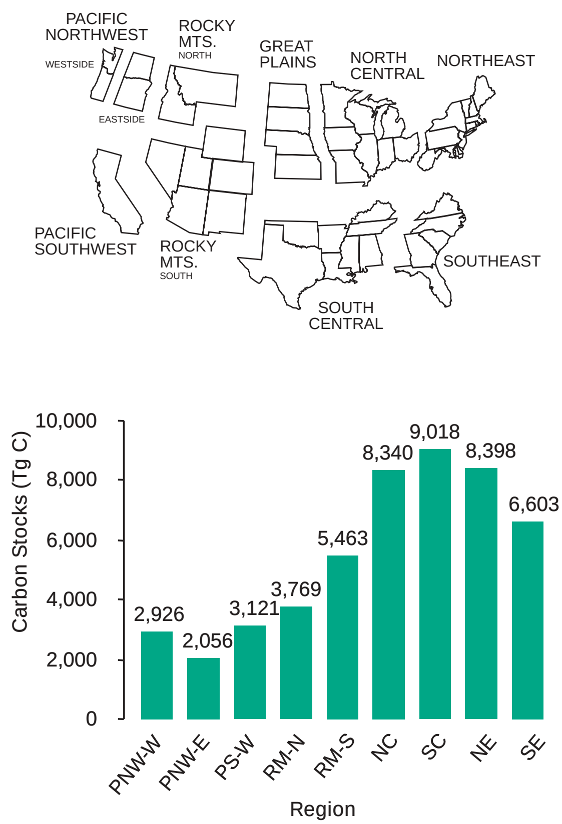
Forest Sector Carbon Pools and Flows



Source: Adapted from Heath and Birdsey (1997)

Figure 6-2

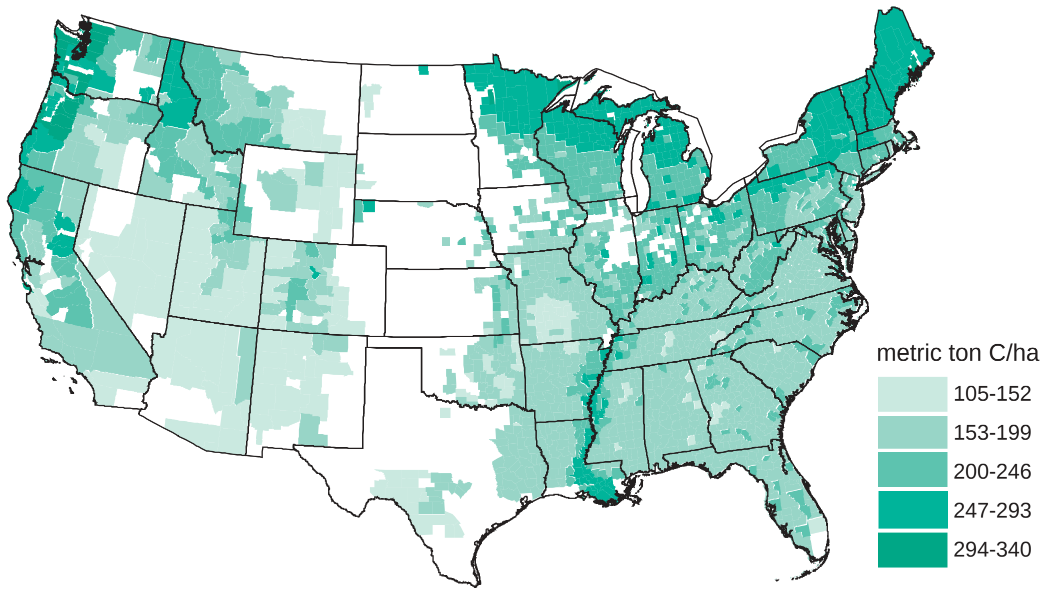
Forest Carbon Stocks by Region, 1997



This graphic shows total forest carbon stocks in 1997, by region. Harvested wood carbon stocks are not included.

Figure 6-3

Forest Carbon Stocks, per Hectare, by County, 1997



This figure shows total, per hectare, forest carbon stocks in 1997, by county. Only counties in the conterminous United States that have at least 5 percent of the county area in forest are shown. Harvested wood carbon stocks are not included.