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A CARBON BUDGET FOR FORESTS OF THE CONTERMINOUS UNITED STATES[1]

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Abstract. The potential need for national-level comparisons of greenhouse gas emissions, and the desirability of understanding terrestrial sources and sinks of carbon, has prompted interest in quantifying national forest carbon budgets. In this study, we link a forest inventory database, a set of stand-level carbon budgets, and information on harvest levels in order to estimate the current pools and flux of carbon in forests of the conterminous United States. The forest inventory specifies the region, forest type, age class, productivity class, management intensity, and ownership of all timberland. The stand-level carbon budgets are based on growth and yield tables, in combination with additional information on carbon in soils, the forest floor, woody debris, and the understory. Total carbon in forests of the conterminous U.S. is estimated at 36.7 Pg, with half of that in the soil compartment. Tree carbon represents 33% of the total, followed by woody debris (10%), the forest floor (6%), and the understory (1%). The carbon uptake associated with net annual growth is 331 Tg, however, much of that is balanced by harvest-related mortality (266 Tg) and decomposition of woody debris. The forest land base at the national level is accumulating 79 Tg/yr, with the largest carbon gain in the Northeast region. The similarity in the magnitude of the biologically driven flux and the harvest-related flux indicates the importance of employing an age-class-based inventory, and of including effects associated with forest harvest and harvest residue, when modeling national carbon budgets in the temperate zone.

Key words: age-class distribution; carbon pools and flux; carbon sources and sinks; forest carbon budget; forest inventory; net ecosystem productivity; regional carbon storage; soil carbon; tree harvest; United States; woody debris.

INTRODUCTION

Rising levels of atmospheric CO_2, and other radiatively important trace gases, are likely to alter the global climate (IPCC 1990, 1992). This concern may lead to international negotiations on carbon emissions, which will require in-depth understanding of national-level carbon budgets. The Framework Convention on Climate Change, signed by the United States and other countries at the June 1992 United Nations Conference on the Environment and Development (UNCED) in Rio de Janeiro, already calls for national inventories of net greenhouse gas emissions (Parson et al. 1992). In many countries, the carbon sources or sinks related to forest management and land use change are of comparable magnitude to carbon emissions from fossil fuel combustion (Post et al. 1990). However, a methodology for evaluating the biological component of carbon flux remains a significant research issue. This study describes an approach to quantifying the pools and flux of carbon associated with forests in the United States.

We have placed strong emphasis on an age-class-based forest inventory in this approach. The importance of an inventory lies in the observation that the sign and the magnitude of net ecosystem productivity changes during the course of succession or stand development (Bormann and Likens 1979, Cropper and Ewel 1984, Sprugel 1985). After a natural disturbance such as catastrophic fire, or human disturbance such as a clear cut harvest, ecosystems are a source of carbon to the atmosphere because of the decomposition of large woody debris and other forms of detritus. Later in stand development, as tree bole volume rapidly accumulates, forest ecosystems are strong carbon sinks. In late succession, rates of tree carbon gain are moderated because greater tree height means more sapwood maintenance respiration and possibly reduced photosynthetic rates (Ryan and Waring 1992, Friend 1993). At the same time, higher mortality may favor accumulation of woody debris. Thus, at the regional or national level, the net annual biological flux of carbon in

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temperate forests depends strongly on the age-class distribution among forest stands.

A second critical aspect of this formulation of a national-level forest carbon budget has been close attention to the rate of harvest and the fate of the harvest residue. In recent decades harvest removals have amounted to a large proportion of current growth in U.S. forests (MacCleery 1992). The merchantable wood removed from the forest and converted into long-lived products may represent a significant sink of carbon. In contrast, the residual tree carbon that is burned, or left on site to decompose, is a potentially large source of carbon to the atmosphere (Houghton et al. 1987, Harmon et al. 1990, Harmon et al. 1991). Previous analyses of forest carbon flux in the U.S. (Armentano and Ralston 1980, Birdsey 1992*a*, Sedjo 1992) have not treated the woody debris component in detail.

The approach to modeling forest carbon flux in this study largely follows that taken in other temperate-zone countries, notably Canada (Kurz et al. 1992), the former Soviet Union (Kolchugina and Vinson 1993), and New Zealand (Maclean and Wakelin 1991). Forest inventory data, in combination with stand-level carbon budgets, and data on harvest levels, are used in those studies to develop forest land base carbon budgets. This approach contrasts with carbon flux estimates based on repeated forest surveys (Birdsey 1992*a*, Kauppi et al. 1992). Studies in tropical countries may require greater emphasis on uneven-aged forests, and on land-use change, which dominates the forest carbon flux in some cases (Houghton et al. 1991*a*, *b*). In general, it is desirable that methodologies used for developing carbon budgets for different countries be standardized if results are to be relevant to the policy community.

METHODS

The basic approach to quantifying the carbon pools and flux was to link a complete forest inventory with a set of stand-level carbon budgets (Fig. 1). The inventory contained information on the area and stocking level for each combination of region, forest type, age class, productivity level, and management intensity. The stand-level carbon budgets contained estimates of carbon in the tree, forest floor, understory, woody debris, and soil at each age class.

The study area covered the conterminous United States (Fig. 2) and we considered all timberland (defined as forest land capable of producing 1.4 $m^3 \cdot ha^{-1} \cdot yr^{-1}$ of industrial wood). Lands available for harvest as well as reserved lands were included. "Other forest land" or woodlands (Waddell et al. 1989) with relatively low biomass and productivity were not considered.

The forest inventory

A forest inventory on private lands in 1990 was provided by the ATLAS inventory projection model (Mills and Kincaid 1992). The inventory was prepared as part of the USDA Forest Service 1989 Resources Planning Act (RPA) Assessment (Haynes 1990) and reflects survey data from regional Forest Inventory and Analysis work units (USDA 1992) and the harvest levels seen in the early 1980s and projected through 1990. The ATLAS inventory includes information on the area and stocking level within each age class for 422 combinations of region, forest type, productivity level, and timber management intensity (e.g., Fig. 3).

Information on the areal extent of forest types on public lands and the total growing stock volume was available from Waddell et al. (1989). Data on the age-class distributions on public lands within each forest type exist for particular areas, but have not been compiled on a national scale. Several assumptions were thus employed for the purposes of this initial carbon budget development. Within the Southeast, South Central, Northeast, and North Central regions (Fig. 2), the area of public timberland is relatively small (Fig. 4). Age-class distribution for public timberlands in these regions was assumed to be equivalent to the age-class distribution on private timberland for the corresponding forest type and region. Total growing stock volume on public lands reported in the 1989 RPA Assessment by forest type and region (Waddell et al. 1989) was allocated among age classes in parallel with the ATLAS inventory data for private lands.

In the Pacific Northwest (West) region, recent state-level studies have collated age-class data for national forests and other public lands including reserved areas (Oregon: Sessions 1991; Washington: MacLean et al. 1991, Adams et al. 1992). This information was used in combination with the Waddell et al. (1989) volumes to estimate age-class distributions and stocking levels. Total carbon data for the oldest age class in the stand-level carbon budgets of this region are comparable to those reported in studies of old-growth forests (e.g., Grier and Logan 1977), so areas of age classes >175 yr were assigned to the 175-yr age class.

In the Rocky Mountain, Pacific Northwest (East), and Pacific Southwest regions, the area of public timberland is quite large; however, the average growing stock volume per unit area is low relative to the Pacific Northwest (West). Lacking comprehensive age-class data, the assumption of equivalent age-class distributions for public and private lands of identical forest type was again invoked by region and reported volumes were again distributed accordingly.

Construction of stand-level carbon budgets

In order to estimate regional and national carbon pools and flux, the carbon pool for each age class in each inventory type was needed. Construction of the stand-level carbon budgets which contained this information began with an ATLAS yield table relating stand age to the volume of growing stock (i.e., merchantable wood). The yield tables used in this analysis were developed as part of the 1989 RPA Assessment (Haynes

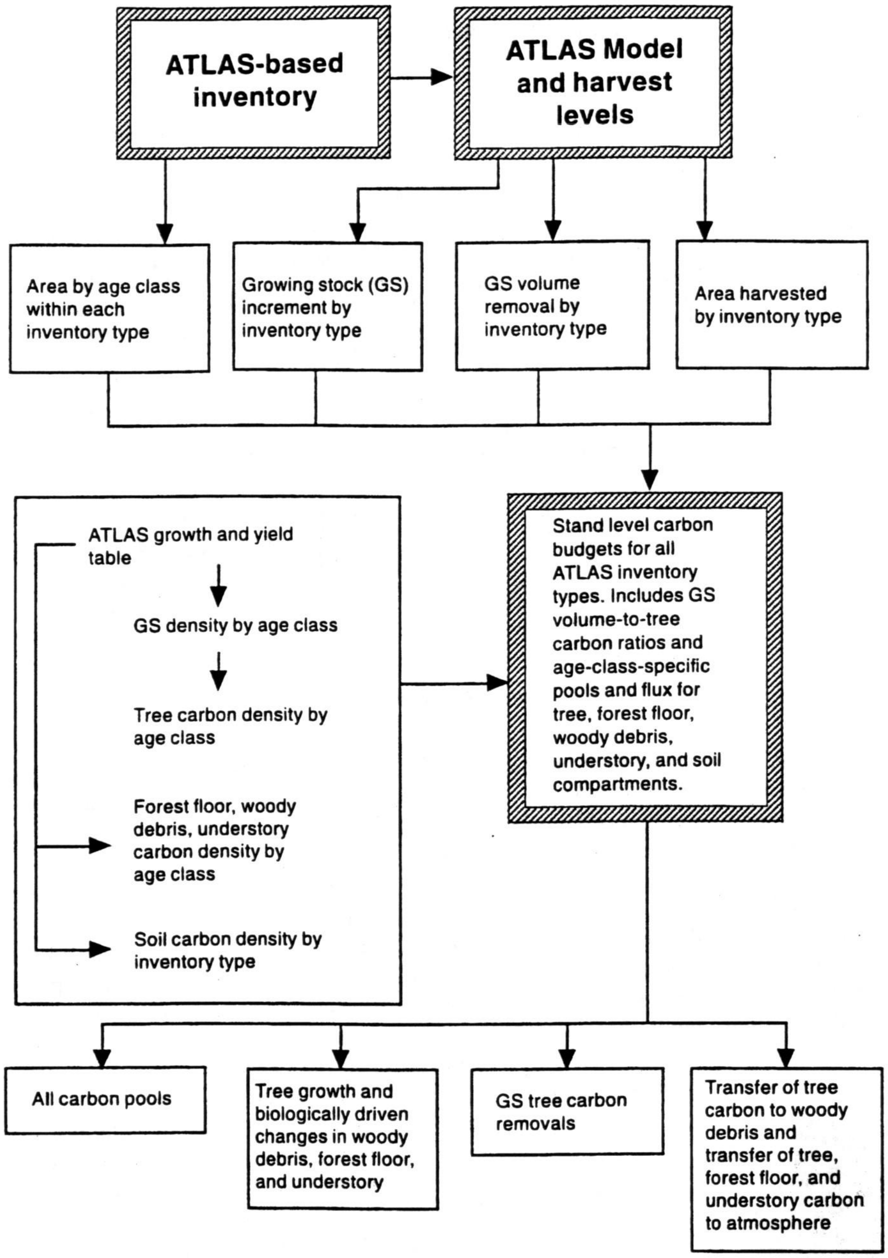

FIG. 1. Data flow diagram for approach to quantifying carbon pools and flux.

1990, Mills 1990). Age-class increments in the tables are 5 or 10 yr, depending on the region. The maximum stand age may reach 90 to 175 yr, depending on region, forest type, and management practices. The yield tables are based on repeated measurements of permanent plots; thus they indicate net growth, i.e., growth minus mortality.

Tree and understory components.—Since the ATLAS yield tables include only growing-stock volume, an initial adjustment factor was applied to account for noncommercial species (1.01 for softwoods, 1.14 for hardwoods). These ratios were based on statistics from selected USDA Forest Service Resource Bulletins (e.g., Thompson 1989). This volume was then converted to bole carbon based on the relative proportion of hardwood and softwood volume and the conversion factors in Table 1. A subsequent conversion from bole carbon to whole-tree carbon was used to account for the contribution of roots, stumps, branches, tops, and cull trees. These ratios (Table 1) were developed from Harmon (1993) and Cost (1990) based on allometric analyses and consideration of forest harvest practices. Lastly, sapling carbon (trees <12.5 cm diameter at breast height [dbh]) was included based on the biomass statistics in the study of Cost et al. (1990). The ratios of sapling carbon to tree carbon were specific to individual

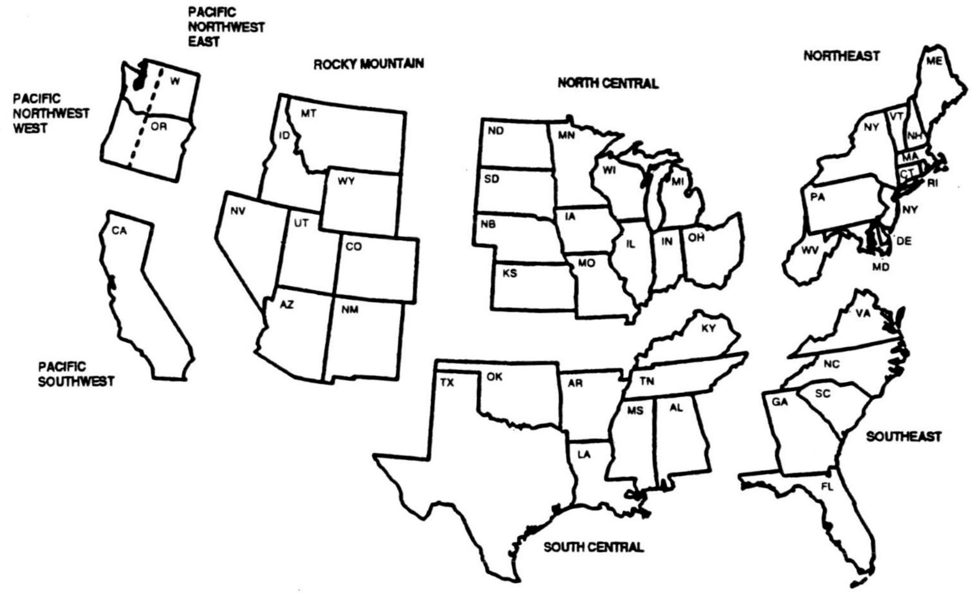

FIG. 2. The conterminous United States study area, divided into regions.

forest types within regions and were dependent on the relative importance of hardwoods and softwoods.

The ratio of whole-tree carbon to merchantable-bole carbon was treated as a constant across age classes within a forest type. The approach here may thus underestimate tree carbon in young stands and overestimate it in old stands (Johnson and Sharpe 1983). Because the absolute volume in young forests is low, the net effect of not accounting for the change in the ratio for those stands is small. In older stands these ratios tend to stabilize, and checks of biomass estimates using the fixed ratio against biomass studies reported in the literature (e.g., Cannell 1982) did not reveal a strong bias.

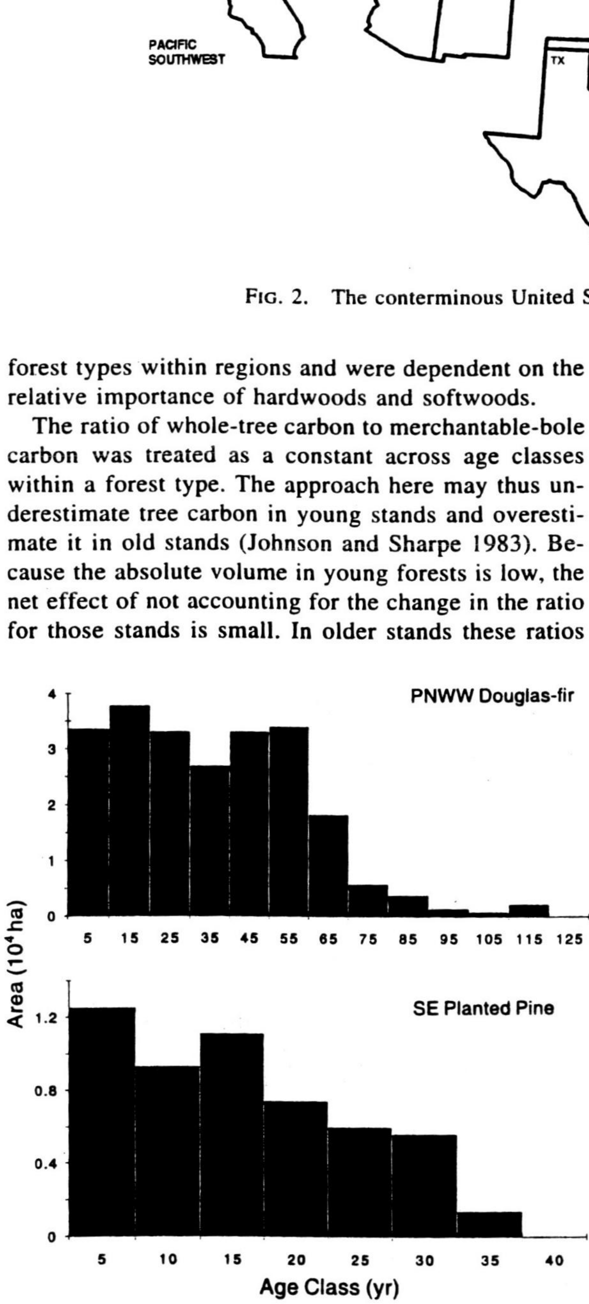

FIG. 3. Stand age-class distribution on private timberland for Pacific Northwest (West) Douglas-fir and Southeast planted pine.

The understory carbon pool consists of herbs and shrubs. The pool sizes were estimated on a forest-type-specific basis and were based on data in the literature (Birdsey 1992*b*). The pool size rises rapidly after harvest or disturbance but falls as the canopy closes and eventually begins rising again as stands mature and gaps occur in the canopy.

Woody debris and forest floor components.—The woody debris pool consists of standing dead trunks, stumps, dead coarse (>2 mm diameter) roots, and dead woody material of >2 cm diameter lying on the forest floor. Because there has been no consistent treatment of woody debris in the USDA Forest Service Forest Inventory and Analysis studies, we used a modeling approach to estimate age-specific woody debris pool sizes.

For the youngest age class, we began with an estimate of the woody debris already on the ground at the time of harvest, and added the residue from live trees left on site after harvest (Harmon 1993). We assumed a whole-bole harvest, average stocking levels, and a standard rotation age. In the Southern regions we assumed that only the roots were left to decay, and the difference between total tree carbon and the sum of roots plus growing-stock carbon was directly emitted. In the other regions 20% of the tree carbon was assumed to be directly emitted and the difference between total tree carbon and growing stock plus direct emissions went to woody debris. The woody debris pools in subsequent age classes were then estimated based on increments due to mortality (a constant applied to

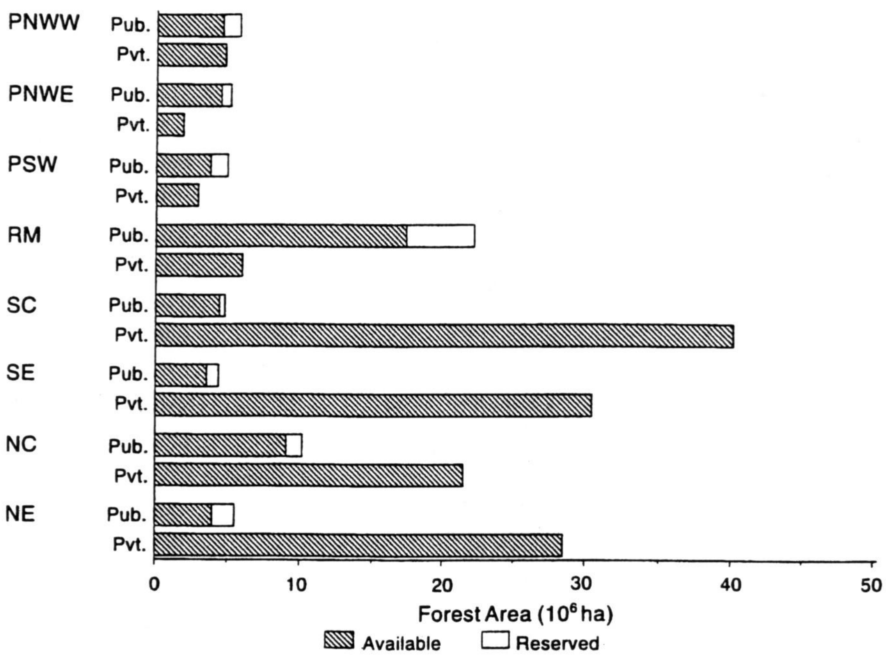

FIG. 4. Distribution of available and reserved timberland by region and ownership. "Reserved" lands are withdrawn from timber utilization by statute or administrative regulation, whereas "available" lands are not. Abbreviations: PNWW = Pacific Northwest (West), PNWE = Pacific Northwest (East), PSW = Pacific Southwest, RM = Rocky Mountains, SC = South Central, SE = Southeast, NC = North Central, and NE = Northeast; Pub. = public, Pvt. = private.

TABLE 1. Factors used in converting growing-stock volume to whole-tree carbon.

		Wood properties			
Region	Forest type*	Specific gravity† (kg/m^3)	Proportion carbon‡ (g/g)	Carbon/ content§ (kg/m^3)	Carbon (whole tree/bole)
Pacific Northwest, Pacific Southwest, and Rocky Mountain	Douglas-fir	473	.512	242	1.54
	Ponderosa pine	416	.512	213	1.82
	Fir–Spruce	349	.512	157	1.69
	Hemlock–Sitka spruce	434	.512	195	1.92
	Lodgepole pine	423	.512	190	1.69
	Larch	508	.512	229	1.54
	Redwoods	416	.512	187	1.47
	Hardwoods	384	.496	172	2.08
North Central	Pines	421	.521	219	1.61
	Spruce–Fir	351	.521	183	1.69
	Oak–Hickory	632	.498	315	1.75
	Maple–Beech	576	.498	287	2.08
	Aspen–Birch	465	.498	232	2.78
	Bottomland hardwoods	758	.498	288	1.64
Northeast	Pines	378	.521	197	1.61
	Spruce–Fir	369	.521	192	1.69
	Oak–Hickory	636	.498	317	1.75
	Maple–Beech–Birch	600	.498	299	2.08
	Bottomland hardwoods	580	.498	288	1.64
Southeast and South Central	Pines	510	.531	271	1.61
	Oak–Hickory	639	.479	318	1.75
	Oak–Pine	639	.497	318	1.67
	Bottomland hardwoods	580	.497	288	1.64

* The first species in a grouped pair is the relatively more important one.

† Weighted average specific gravity of the three most common (in terms of volume) softwood or hardwood species within the forest type (Birdsey 1992*a*).

‡ From Birdsey (1992*a*).

§ The reported values for specific gravity and percent carbon apply to the dominant fiber type within the forest type. When forest types contained a mixture of hardwood and softwood species the relative contribution of each was accounted for in this final ratio of carbon content to wood volume.

TABLE 2. Constants (yr^{-1}) for decomposition and mortality used in modeling of woody debris dynamics.

Region	Forest type	Decomposition	Mortality
Pacific Northwest	Douglas-fir	.018	.0050
	Ponderosa pine	.015	.0050
	Fir–Spruce	.024	.0050
	Hemlock–Sitka spruce	.029	.0060
	Lodgepole pine	.036	.0045
	Redwoods	.012	.0021
	Hardwoods	.067	.0060
Rocky Mountain	Douglas-fir	.018	.0023
	Ponderosa pine	.015	.0024
	Spruce–Fir	.014	.0062
	Larch	.017	.0040
	Lodgepole pine	.022	.0040
North Central	Pines	.035	.0058
	Spruce–Fir	.035	.0100
	Oak–Hickory	.052	.0037
	Maple–Beech	.072	.0066
	Aspen–Birch	.067	.0147
	Bottomland hardwoods	.098	.0143
Northeast	Pines	.035	.0058
	Spruce–Fir	.035	.0100
	Oak–Hickory	.067	.0075
	Maple–Beech–Birch	.057	.0066
Southeast, South Central	Natural pines	.042	.0119
	Planted pines	.049	.0119
	Oak–Hickory	.067	.0074
	Oak–Pine	.052	.0097
	Bottomland hardwoods	.098	.0081

the whole tree pool) and decrements due to decay (a constant applied to the woody debris pool). The mortality and decay constants (Table 2) were from the study of Harmon (1993) and reflected data in the literature (e.g., Harmon et al. 1986). The mortality estimates are based on USDA Forest Service permanent plots and include death by all natural causes, including fire. The total mortality for the U.S. using this approach was approximately the same as reported in Waddell et al. (1989).

TABLE 3. Mean soil organic carbon by forest type.

Forest types	Mean soil carbon density (kg/m^3)
Western U.S.	
Pinyon–Juniper	6.5
Chaparral	7.0
Redwood	7.5
Ponderosa pine	8.1
Hardwoods	8.3
Larch	8.7
Western white pine	8.7
Lodgepole pine	8.8
Fir–Spruce	9.4
Douglas-fir	10.2
Hemlock–Sitka spruce	14.3
Eastern U.S.	
Oak–Pine	6.8
Loblolly pine–Shortleaf pine	7.9
Oak–Hickory	8.3
Oak–Gum–Cypress	10.7
Longleaf pine–Slash pine	11.3
Elm–Ash–Cottonwood	11.7
Aspen–Birch	13.1
White pine–Red pine–Jack pine	13.3
Maple–Beech–Birch	13.8
Spruce–Fir	15.1

The modeled levels of woody debris were generally consistent with the limited observational data on woody debris in natural and managed stands (Harmon et al. 1986, Agee and Huff 1987, Mattson et al. 1987, Spies et al. 1988, Muller and Liu 1991, Means et al. 1992). This modeling approach does not take into consideration the complex history (MacCleery 1992) nor ongoing changes (Swanson and Franklin 1992) in forest management and fire regimes in the U.S., but it is a start towards accounting for an important component of total forest carbon (Harmon et al. 1986, Harmon et al. 1991).

Estimates of initial forest floor carbon and the age-specific increases in its pool size were based primarily on Vogt et al. (1986) and are the same as used in Birdsey (1992*b*). The pool size was adjusted in proportion to the stocking level in the current inventory.

Soil component.—To arrive at representative soil carbon pools for each forest type (Table 3), geographic information systems software was used to overlay the spatial distribution of each forest type with a map of soil carbon. The soil carbon map was developed by Kern (1994). In that study, data from over 5000 pedons on record at the U.S. Soil Conservation Service Lincoln Laboratory (Lincoln, Nebraska) were used to estimate mean soil carbon at the great group taxonomic level (USDA 1975). The geographic distribution of the great

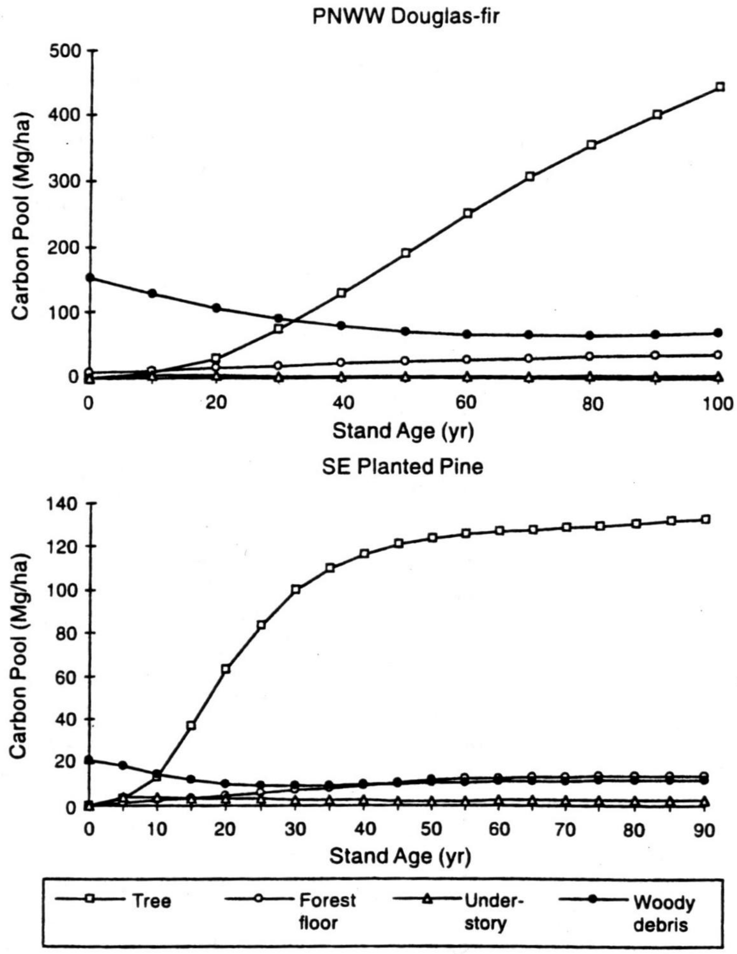

FIG. 5. Examples of stand-level carbon budgets for two forest types.

groups was derived from the 1982 National Resource Inventory (SCS 1987). Forest-type distributions for the soil carbon analysis were taken from a digitized version of the Society of American Foresters map of the major forest types (Eyre 1980). The map was developed from indicators of potential vegetation and USDA Forest Service inventory data. The spatial resolution of the forest-type map was relatively coarse and necessitated omitting from the averaging scheme some peatland areas (24×10^6 ha) in the North Central region where the spatial heterogeneity in soil carbon may have been high. Studies of soil carbon distribution in relation to climatic factors have found that soil carbon generally increased with cooler and wetter environments (Post et al. 1982). The estimates of soil carbon for these forest types and the climatic regimes associated with each forest type (Burns and Honkala 1990) appear to be consistent with this pattern. Average soil carbon in the North Central region compared closely with the regional estimate in Grigal and Ohmann (1992).

Possible changes in soil carbon storage over the course of stand development could depend on many factors related to changing rates of carbon input, decomposition, leaching, and erosion. Earlier carbon models designed for analysis of land-use change over large geographical areas (e.g., Moore et al. 1981) have used a 20% or more decrease in the "soil" carbon pool after harvest. However, the soil pool was actually composed of soil, litter, and woody debris in those models. A recent review of studies in temperate forests did not indicate a significant decrease in soil organic carbon after tree harvest (Johnson 1992). In the case of conversion from cropland or pasture to forest, there is likely to be a slow increase in soil carbon storage (Jenkinson 1971), but these possible changes have not been well quantified. Considering the uncertainty in the temporal pattern to be expected, we have held soil carbon constant in these stand-level carbon budgets.

The general pattern for the change in carbon pools during stand development was similar across forest types (Fig. 5). Immediately after harvesting, woody debris is the largest pool. After one to two decades, woody debris has declined and the tree carbon pool has surpassed it. The most rapid accumulation of tree carbon occurs in the Southeast region, but the maximum tree carbon storage is lower there than in other regions, especially the Pacific Northwest (West).

Carbon pools and flux

Carbon pools were quantified by combining data on the areal extent and stocking level of each age class, within each inventory type, with the carbon pool values in the associated stand-level carbon budget.

The reference state in the carbon flux analysis was considered the net accumulation or loss of carbon from the forest land base over a 1-yr period. The term "flux" thus includes carbon transfer between the forest and the atmosphere in either direction and carbon transfer out of the forest for human use. In an effort to isolate the effects of harvesting, we have distinguished between biologically driven carbon flux and harvest-driven carbon flux.

Biologically driven carbon flux.—The biologically driven carbon flux is an estimate of the change in carbon storage expected over a 1-yr period assuming there had been no harvest. The processes involved are primarily photosynthesis, respiration (including heterotrophic), and tree mortality. This flux for individual forest components, and total flux that reflects net changes in all the pools (i.e., net ecosystem production), was based on the inventories and stand-level carbon budgets described earlier. The estimate of net carbon uptake or loss on an annual basis for a given age class was calculated as the change in the carbon pool over the ensuing time interval, divided by the number of years in the interval (e.g., 5 yr in the Southeast region and 10 yr in the Pacific Northwest). The effect of stocking level on tree growth was manifest as a small amplification in tree growth in the case of low stocking levels to reflect higher potential growth rates. These growth effects are prescribed in ATLAS (Mills and Kincaid 1992).

The approach used here does not account for woody debris decomposition on lands that have been deforested and are no longer in the forest inventory. This factor may not be large in the U.S., which has a relatively stable forest land base. However, in regions or countries of rapid deforestation, the forest inventory should, in principle, carry lands deforested in previous decades and include this source of carbon.

Harvest-driven carbon flux.—The average volume of harvest-related growing stock reduction for the 1990s is from the projection associated with the 1989 RPA Assessment. These removal rates are comparable to those reported in Powell et al. (1993) although somewhat higher (<10%) because they represent the average for the decade. The distribution of the harvest among forest types and age classes for private lands in the 1990s was derived from the ATLAS model. The algorithms employed by ATLAS for selecting age classes, such as "oldest first", are based on conventional harvest practices. Information on land-use change was also derived from the ATLAS model run associated with the 1989 RPA Assessment. In that projection the average rate of reduction in the forest land base in the 1990s was 0.16×10^6 ha/yr (Alig et al. 1990).

In order to estimate the area harvested on public lands and its distribution among forest types and age classes, we used the RPA Assessment volumes and generally assumed an equivalence in the distribution of harvest across forest types and age classes between public and private lands. In the forests of the Pacific Northwest region we used data from Sessions (1991) on the distribution of the harvest among age classes on public lands.

The tree carbon on harvested lands that was not accounted for as growing-stock removals or formation of woody debris residue (calculated as the difference between post-harvest and pre-harvest woody debris on all harvested areas) was assumed to be emitted within the year of harvest. This residual could be considered to include slash burns and perhaps additional material that might be removed from the site but returned to the atmosphere relatively quickly (e.g., fuelwood). The understory and part of the forest floor of harvested stands were also assumed to be either burned or rapidly decomposed, and this loss was likewise treated as instantaneous.

Fire emissions.—A separate estimate of forest wildfire emissions is not a component of this carbon budget since all mortality derived from wildfire is assumed to be transferred to the woody debris pool. This approach results in slower release of carbon to the atmosphere than would be the case with modeling direct emissions, but it ensures that the budget accounts for all carbon associated with mortality. Direct wildfire emissions appear to be on the order of 20 Tg/yr, using a long-term average (A. Auclair [Science and Policy Associates, Washington, D.C., USA], *personal communication*). Approximately one third of the wildfire emissions are associated with woodlands, which tends to maintain a long-term carbon equilibrium on those lands. Note that prescribed fire, including slash burns, may generate an additional 10–20 Tg/yr (Yamate 1974); however, this flux is also captured in our modeling of harvest-driven carbon flux. Formation of charcoal has not been considered in this analysis.

Results and Discussion

The total land of the conterminous U.S. is 765.5×10^6 ha of which 200.7×10^6 ha is timberland. Woodland, characterized by production <1.4 $m^3 \cdot ha^{-1} \cdot yr^{-1}$, comprises another 42.3×10^6 ha, the majority of which is in the West. A check of forest area derived from this inventory, with forest cover derived from a recent land cover map based on satellite remote sensing (Loveland et al. 1991), indicated reasonable agreement (within 4%) at the national level (Turner et al. 1993). At the state and regional levels, greater differences were apparent.

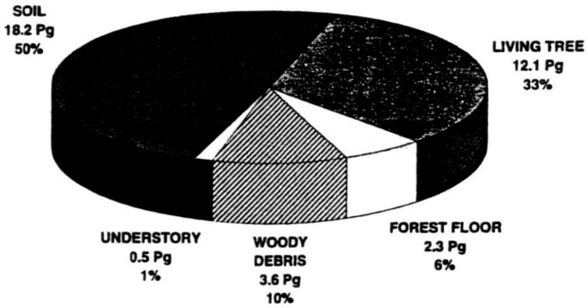

FIG. 6. Relative contribution of forest components to total carbon on U.S. timberland.

Carbon pools

Half of the total timberland carbon is in the mineral soil (Fig. 6). Tree carbon, which includes coarse roots, is the next largest component at 33%, followed by woody debris (10%), forest floor (6%), and understory (1%). Total carbon storage in living trees on timberland in the U.S. is estimated at 12.1 Pg and average tree carbon storage was 6.1 kg/m^2. The total tree biomass value is ≈5% greater than a previous estimate by Birdsey (1992*a*).

In terms of the regional distribution, the forests in the Northeast and South Central regions have the largest absolute quantities of carbon (Fig. 7). A different pattern is seen for the average quantity of carbon per unit area, with the highest average (33 kg/m^2) found in the Pacific Northwest (West) and the lowest (15 kg/m^2) in the South Central region (Fig. 8). Woody debris accounts for a significant proportion (21%) of the carbon in the Pacific Northwest, but less so elsewhere.

Carbon flux

Biologically driven flux.—The estimates of carbon flux derived from the stand-level carbon budgets indicate a common sequence over the course of stand development (Fig. 9). After a disturbance, the net ecosystem productivity (NEP), or net change in the total carbon pool, is low or negative because of carbon loss to the atmosphere primarily associated with decomposition of the woody debris pool. Near the time of canopy closure, the rate of carbon accumulation associated with tree growth has usually begun to exceed the rate of carbon emissions from woody debris, and the system as a whole is a carbon sink. For older age classes the NEP decreases again because greater maintenance respiration costs and other physiological or anatomical constraints decrease growth in the living-tree pool. This effect is somewhat compensated for in late stand development by the increase in woody debris carbon. The time course of the fluctuations in NEP varies with forest type. NEP in the Southeast planted-pine forest type peaks much earlier than in the Pacific Northwest Douglas-fir forest type.

The flux of carbon into the forest from net tree growth is 331 Tg/yr. Carbon also accumulated in the understory (6 Tg/yr) and forest floor (33 Tg/yr) pools. The only pool to consistently show a net carbon emission (62 Tg/yr) is the woody debris pool (Fig. 10). This pattern is driven by the decay of woody debris in the early to middle stages of stand development. During these periods, the quantity of carbon emitted from woody debris created at the time of stand origin (due to harvest, wildfire, etc.) and from woody debris remnant from the previous stand, far exceeds inputs to the

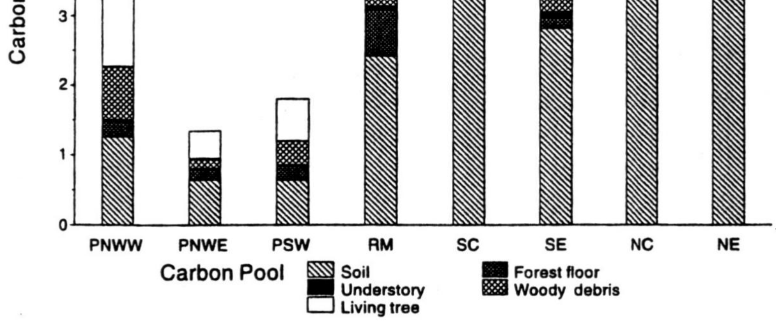

FIG. 7. Carbon pools by region and compartment for U.S. timberland. Region abbreviations are as in Fig. 4.

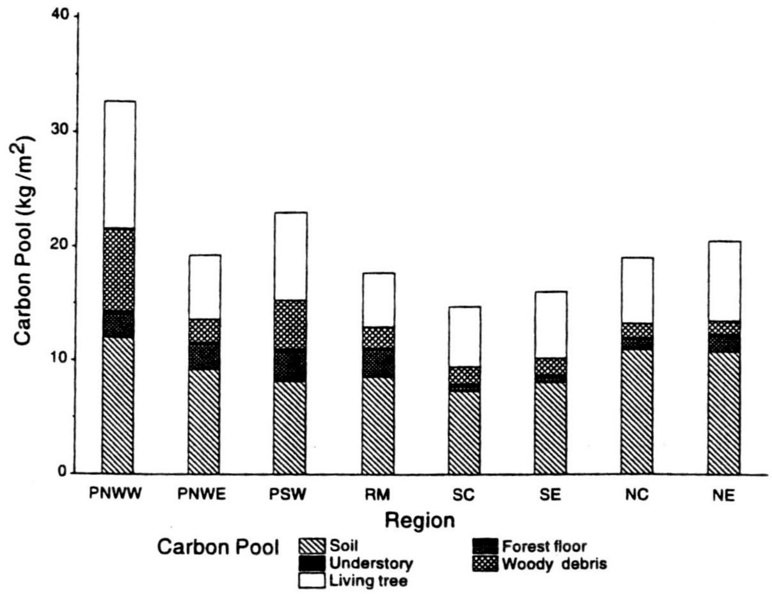

FIG. 8. Average carbon storage per unit area by region and component. Region abbreviations are as in Fig. 4.

compartment originating in tree mortality. Woody debris decomposition provides the greatest offset to tree growth in the Pacific Northwest (West) and has a minor impact in the eastern regions (Fig. 11).

The overall net flux of carbon driven by biological processes is 308 Tg/yr moving from the atmosphere into forest stands. The two southern regions have the highest potential accumulation (Fig. 10) followed by the two northern and four western regions. The average net uptake per unit area (Fig. 11) is highest in the Southeast, South Central, and Pacific Northwest (West) regions. The North Central, Northeast, and Pacific Southwest regions have intermediate levels of net uptake, and the Rocky Mountain and Pacific Northwest (East) regions have the lowest uptake per unit area, reflecting relatively dry and cold climates.

In the Pacific Northwest (West), where the age-class distribution on public lands was taken into consideration, private lands accounted for 65% of the net uptake but only 45% of the total timberland area. That dif-

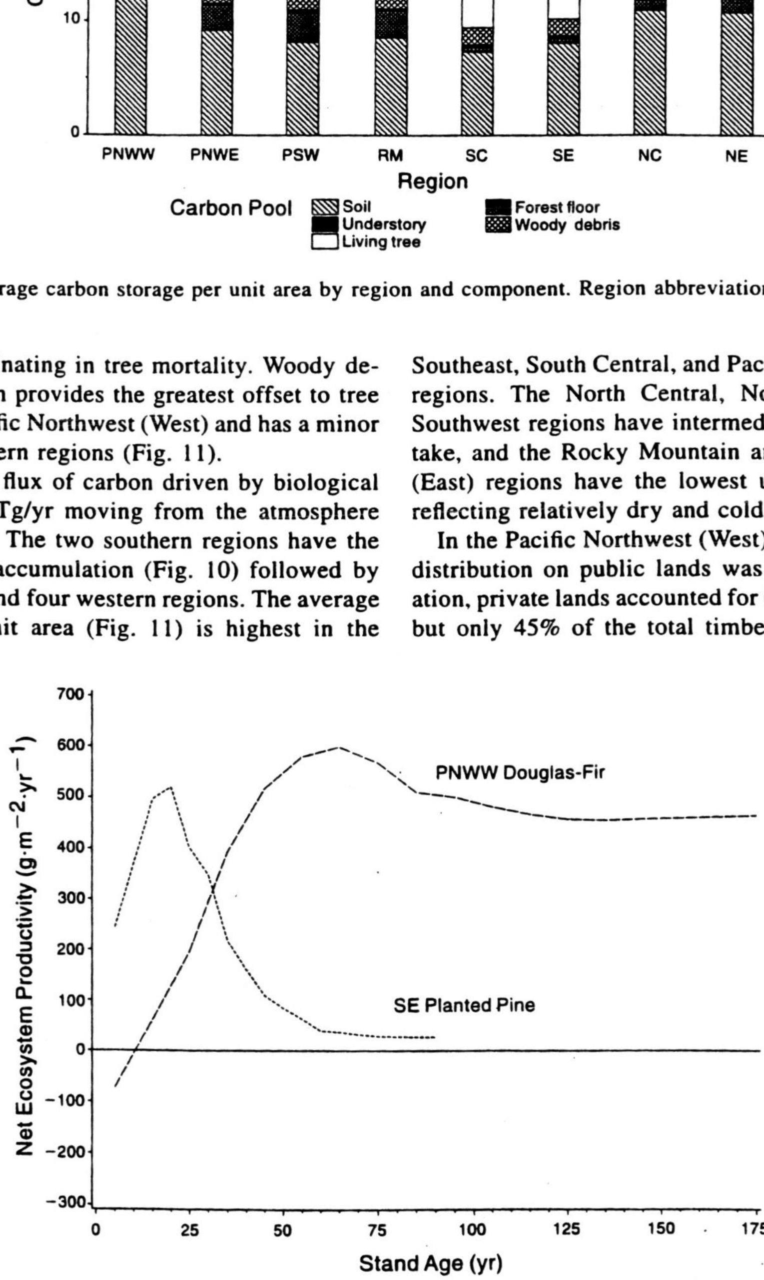

FIG. 9. Net ecosystem productivity during stand development for two forest types. Positive values are carbon transfer from the atmosphere to the forest land base. Negative values are carbon transfer to the atmosphere. Region abbreviations are as in Fig. 4.

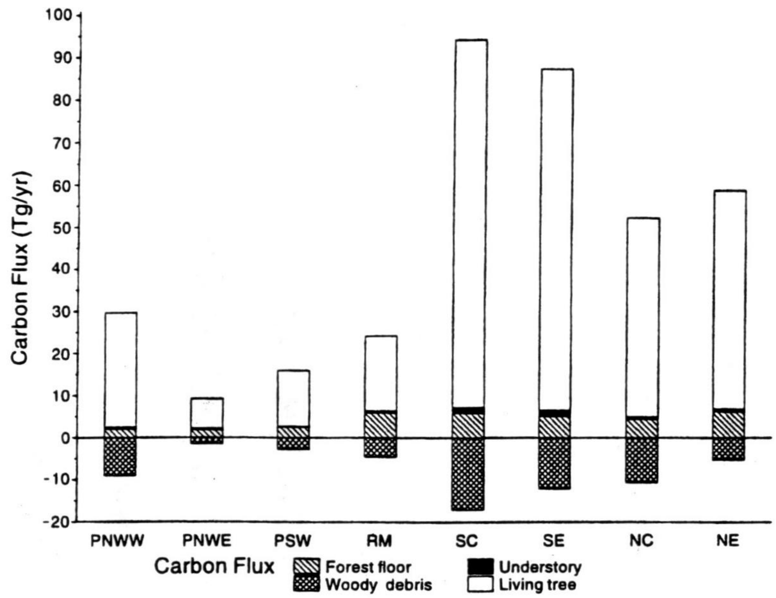

FIG. 10. Carbon flux by region and component. Positive values are net carbon transfer from the atmosphere to the forest land base. Negative values are net carbon transfer to the atmosphere. Region abbreviations are as in Fig. 4.

ference is due to the greater productivity of the younger stands, which characterize private lands in this region. Sessions (1991) reported that 40% of the total area of public timberland in Oregon was >150 yr of age, while the comparable value for Douglas-fir stands on forest industry lands (based on the RPA inventory) was ≈5%.

Limited opportunities exist for validation of the biological flux estimates. Powell et al. (1992) estimated timber volume growth on U.S. available timberland at 0.59×10^9 m^3 for 1991 based on repeated sampling of permanent plots. The comparable value in the present analysis is the same, although the estimate here is an average for the 1990s. The tree growth estimate here (331 Tg/yr) also compares closely with the estimate in Birdsey (1992*a*).

Harvest-driven carbon flux.—The annual carbon reduction in growing stock associated with harvest amounted to 124 Tg/yr, with the largest quantities occurring in the South Central and Southeast regions (Fig. 12) and with public lands accounting for 17% of the

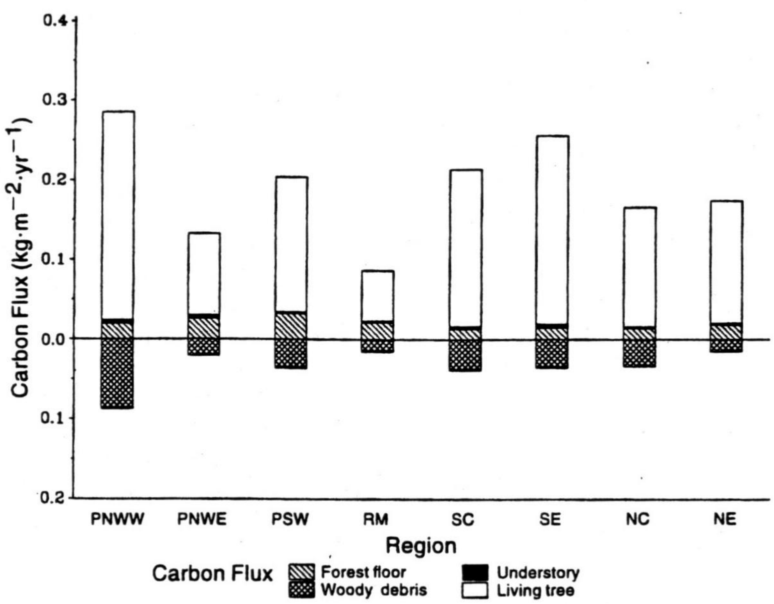

FIG. 11. Average biologically driven carbon flux by region and component. Positive values are net carbon transfer from the atmosphere to the forest land base. Negative values are net carbon transfer to the atmosphere. Region abbreviations are as in Fig. 4.

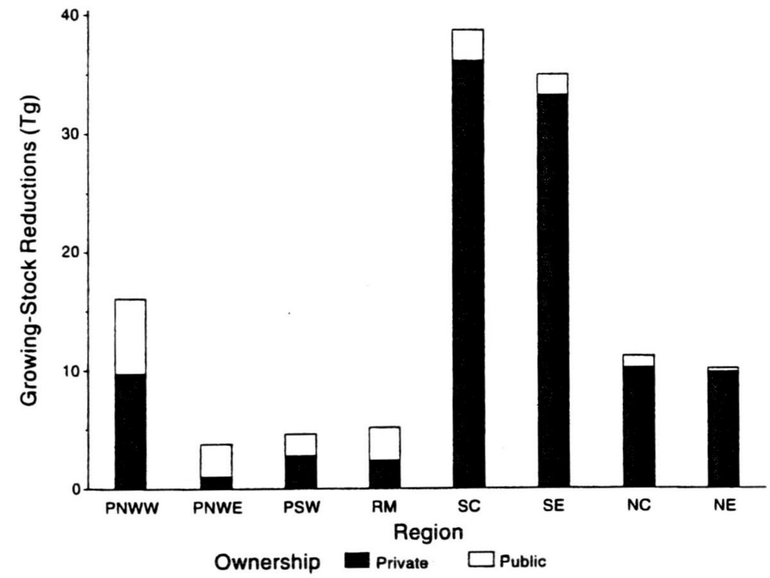

FIG. 12. Growing-stock reductions associated with harvesting. Region abbreviations are as in Fig. 4.

total. The area harvested is estimated at 2.85×10^6 ha. The effect of timber harvest on the carbon budget is manifest in several ways. Most significant is the transfer of a large quantity of carbon out of the live tree biomass pool (266 Tg). This tree carbon represents both the growing-stock and nongrowing-stock components of the tree carbon on the harvested lands. In the U.S. this reduction in tree carbon is 80% of the annual carbon accumulation from tree growth. The difference between annual growth and harvest removals is greatest in the Northeast region for private lands and in the Rocky Mountain region for public lands.

The nongrowing-stock component of the tree carbon on the harvested land (142 Tg/yr) is partitioned between the harvest emissions (72 Tg/yr) and a transfer to the woody debris pool (70 Tg/yr). Partitioning of the harvest residue between formation of woody debris (49%) and short-term emissions (51%) is consistent with the observation that roots and stumps (≈20% of tree carbon) generally remain after harvest.

The rapid change in forest floor and understory carbon associated with or immediately following harvesting amounted to a 33 Tg/yr loss of carbon, bringing the total harvest emissions to 105 Tg/yr.

TABLE 4. Carbon balance on the forest land base. "Biological" and "Harvest" refer to the net change in pool size as a function of biologically related or harvest-related factors.

	Tree (Tg/yr)	Woody debris (Tg/yr)	Forest floor (Tg/yr)	Understory (Tg/yr)	Net (Tg/yr)
Biological	331	−62	33	6	308
Harvest	−266*	70	−25	−8	−229
Combined	65	8	8	−2	79

* Includes growing-stock removals (124 Tg/yr), woody debris formation (70 Tg/yr), and harvest emissions (72 Tg/yr).

Complete base-year flux analysis.—The net effect of forest growth and microbial decomposition is a transfer of 308 Tg/yr of carbon from the atmosphere into organic matter. This rate of potential accumulation is possible only because the removal of carbon by harvesting has tended to maintain a large area in the early to middle stages of stand development, which favor carbon accumulation. Under unmanaged conditions the rate of carbon accumulation on the same land base would be lower or even negative, because fire and increased rates of woody debris decomposition would balance uptake. The potential accumulation of 308 Tg/yr is partially offset by harvest-related carbon removals from the forest land base totaling 124 Tg/yr and harvest emissions of 105 Tg/yr. The difference between the biological flux and the losses associated with harvest is a carbon gain of 79 Tg/yr on the forest land base (Table 4).

The live tree carbon pool is increasing because tree growth is greater than the carbon in mortality from natural causes and harvesting. The simplicity of the assumptions about woody debris dynamics lends a high degree of uncertainty to the net flux estimate for woody debris (+8 Tg/yr). Over the last several decades, utilization of harvest residues has tended to increase and the inventory of older forests carrying large quantities of woody debris has decreased. At the same time, the average age-class of stands in the U.S. has probably fallen because of increasing harvesting and management. Within a management cycle, younger stands may carry more woody debris in the form of stumps and harvest residues. These counteracting factors make it difficult to assess the overall change in the storage of woody debris. More recently, forest management regulations have begun to require that some woody debris be left on site for wildlife habitat, and that trend will also favor woody debris accumulation.

The results here for net accumulation of tree carbon are comparable with other recent national analyses. Heath and Birdsey (1993) used current inventory data and inventory projections to estimate accumulation of live and dead carbon for timberland in the conterminous U.S. The live carbon accumulation for the period 1987–2010 was 17 Tg/yr. That value reflects the general trend of a decreasing carbon sink on the forest land base over the coming decades (Turner et al. 1993) and does not include reserved timberland, which we estimate is sequestering ≈10 Tg/yr. Our estimate of tree carbon gain for the 1990s is also roughly consistent with the growing-stock accumulation for 1991 reported in Powell et al. (1993). They indicated an annual accumulation (net growth − removals) on available timberland of ≈140×10^6 m^3 of growing stock, which represents ≈34 Tg of growing stock or ≈70 Tg of whole-tree carbon. Subak et al. (1993) similarly estimated the 1990 carbon accumulation on the forest land base at 60 Tg/yr based on inventory statistics from the Food and Agriculture Organization of the United Nations.

The flux of dead carbon has only been estimated by Heath and Birdsey (1993), whose assumptions about soil and woody debris differ significantly from ours. The fate of nongrowing-stock tree carbon after harvest and woody debris decomposition in general are not treated explicitly in their analysis. Soil carbon is assumed to decline 20% after harvest and return to the preharvest level over the course of one rotation. Lands that have reverted from agriculture or pasture to forest are also considered to be a carbon sink. The net effect is a dead carbon sink of 55 Tg/yr over the period 1987–2010 (Heath and Birdsey 1993), which contrasts with our estimate of a woody debris sink of 8 Tg/yr and a stable soil carbon pool. A large soil carbon sink does not appear to be consistent with the observation that the forest land area of the U.S. has been stable over the last 50 yr (Alig et al. 1990, MacCleery 1992), so that soil carbon gains from reversion of nonforest land to forest would be balanced to some degree by losses from conversion of forest land to other uses.

Regional trends.—The carbon sink at the national level masks significant regional differences. The largest net sink is in the Northeast region (31 Tg/yr). This sink reflects, in part, the reversion of lands from marginal agriculture into forests (Williams 1988) and the lack of demand for the hardwood resource (Haynes 1990). There has also been substantial land abandonment in the Southeast and South Central regions earlier in this century (Hart 1980, Sharpe and Johnson 1981), and estimates of a regional carbon sink on the order of 70 Tg/yr have been presented (Armentano and Ralston 1980, Harris and Delcourt 1980, Schiffman and Johnson 1989). However, timberland area has decreased in the South over the last few decades, primarily as a function of converting bottomland hardwoods to agriculture (Alig et al. 1990).

In the Rocky Mountains the land base is stable and rates of growth and harvest are relatively low, so little change in carbon storage is evident. The Rocky Mountain region has a large area of reserved timberland on which growth is slow, but carbon is probably accumulating.

For the Pacific Coast states, private lands have nearly completed the transition from primary forest to managed secondary forests and are approaching carbon equilibrium. On public lands, the trend into the early 1990s was reduction in old-growth forests on the lands available for harvest (Haynes 1990). This transition to managed secondary forests tends to decrease the live biomass in the region and create a carbon source associated with decay of woody debris (Harmon et al. 1990).

The policies related to management of old-growth forest have recently been changing in response to public concerns about habitat preservation. Harvest levels on public lands in the Pacific Northwest are decreasing and harvest methods that promote retention of live trees and woody debris (Swanson and Franklin 1992) are being implemented. In a simulation based on the carbon budget in the present study, a 20% reduction in the harvest on National Forest land produced an additional carbon sink of >3 Tg/yr (Turner et al. 1993). Much greater reductions are being considered, so the carbon sink on public lands is expected to increase.

Temperate zone carbon sink

As a whole, the U.S. is following the same trend as many northern temperate-zone countries (Clawson 1979, Armentano and Ralston 1980, Kauppi et al. 1992, Sedjo 1992, Birdsey et al. 1993). Recovery from earlier periods of extensive forest harvest and limited management is now resulting in carbon accumulation. Specific factors in the U.S. include reversion of agricultural land to forest (Williams 1988) and fire suppression (MacCleery 1992). The combined effects of these factors in northern temperate-zone forests is believed to be an accumulation of 700 Tg/yr (Sedjo 1992). A terrestrial sink of this magnitude is consistent with the CO_2 tracer model of Tans et al. (1990) and helps account for a carbon sink needed to balance the global carbon budget (Post et al. 1990, Siegenthaler and Sarmiento 1993).

Temperate-zone countries may not continue accumulating carbon for long. Anywhere that harvests or forest degradation exceed growth rates, carbon storage on the forest land base will decrease. In Canada, the tree carbon pool is decreasing because the sum of harvest removals and natural losses exceeds growth rates (Kurz et al. 1992). In the U.S., projections call for a 5% loss in the private timberland area by the year 2040 (Alig et al. 1990). A general intensification of forest management, resulting in lower carbon storage per unit area (Cooper 1983, Dewer 1991), and a gradual increase in the harvest level (Haynes 1990), are also expected. These factors will tend to militate against a stable or increasing carbon sink (Turner et al. 1993). Increasing temperatures, atmospheric CO_2, and nitrogen deposition could promote higher growth rates (McGuire et al. 1993), but projected climate change is also likely to produce a transient release of forest carbon because carbon sources associated with increasing disturbance rates would be greater than carbon sinks associated with land recovering from disturbance (King and Neilson 1992).

A variety of studies are needed to reduce uncertainties in the present analysis. A better forest inventory for public lands would increase confidence in the growth estimates on those lands. Satellite remote sensing has been employed in development of forest inventories and may be particularly effective in the coniferous forests of the western U.S. (Teply and Green 1991, Cohen et al. 1992). The loss or gain of soil organic matter after harvesting and later in stand development is also poorly understood (Johnson 1992). Even a small soil-carbon accumulation expressed over a large area results in a significant carbon sink. The pools and flux of woody debris are also highly uncertain and will benefit from additional observations during forest surveys and from a more complex modeling approach that accounts for stand history.

Conclusions

Increasing attention will be paid to national forest-sector carbon budgets because of the Framework Convention on Climate Change (Parson et al. 1992). Each country has a different forest age- or size-class distribution, as well as different harvest levels and trends in land-use change. Nevertheless, a common framework for quantifying carbon flux is desirable. This study emphasizes the importance of an age-class-based inventory, a set of stand-level carbon budgets that treat living as well as non-living carbon pools, and close attention to the magnitude of forest harvest.

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