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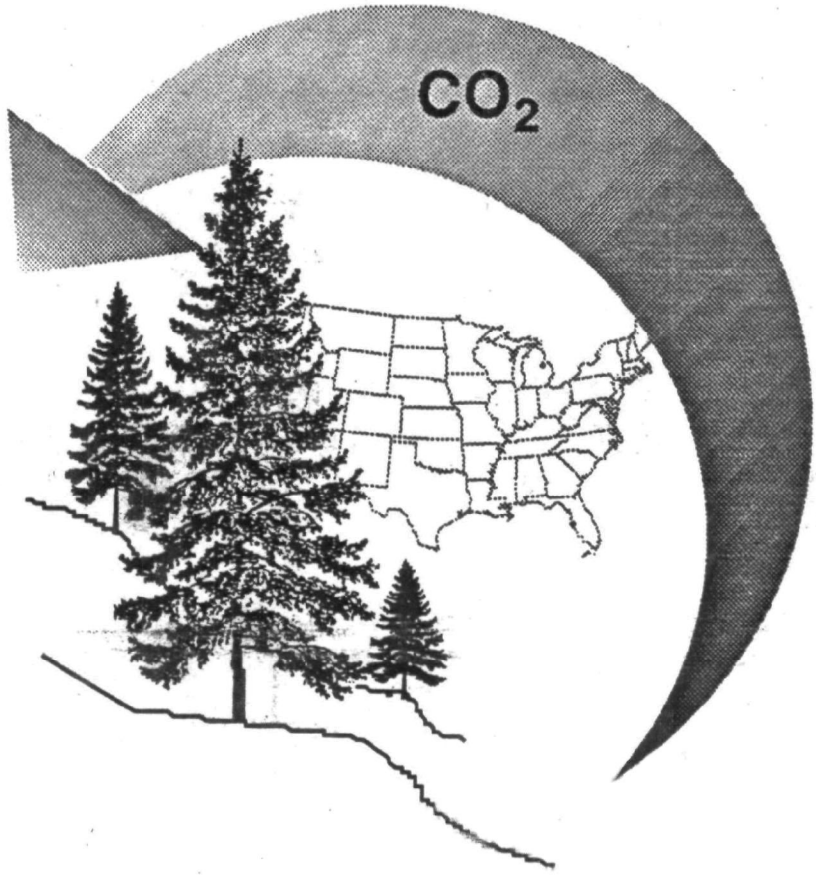

THE FOREST SECTOR CARBON BUDGET OF THE UNITED STATES: CARBON POOLS AND FLUX UNDER ALTERNATIVE POLICY OPTIONS

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Unit Conversions

Documents treating the relationship of policy issues to the global carbon cycle typically use units of megatons (Mt) and gigatons (Gt). We have used these units in the executive summary but throughout the remainder of this report we use primarily SI units. Some of the relevant conversions are as follows:

$$\text{Mt} = \text{megaton} = \text{million metric tons} = 10^{12}\ \text{g} = \text{teragram} = \text{Tg},$$

$$\text{Gt} = \text{gigaton} = \text{billion metric tons} = 10^{15}\ \text{g} = \text{petagram} = \text{Pg}.$$

The pools and flux of biomass or carbon dioxide are expressed as carbon throughout the report.

Executive Summary

Rising levels of atmospheric CO_2 and other radiatively important trace gases (e.g., CH_4, N_2O) from anthropogenic sources are likely to induce changes in the earth's climate over the coming decades (Houghton et al. 1991). While there remains considerable uncertainty about the rate and magnitude of the possible climate change, there is an emerging consensus that policies relevant to stabilizing or reducing the emission of CO_2 and other "greenhouse gases" should be explored (NAS 1991, Rubin et al. 1992).

Terrestrial ecosystems have a major role in the global cycling of carbon. Vegetation and soil contain about three times as much carbon as is in the atmosphere. Also, each year, they exchange about 20 times as much carbon with the atmosphere as is emitted from fossil fuel use. For this reason, one proposed approach for slowing the increase in atmospheric CO_2 is to manage terrestrial ecosystems to conserve and sequester carbon by changing the balance between the uptake of carbon dioxide from the atmosphere through photosynthesis and the release of carbon dioxide through respiration and biomass burning (Schroeder and Ladd 1991; Sedjo and Solomon 1989; Sampson and Hair, in press). Forest ecosystems are particularly important in consideration of carbon conservation and sequestration opportunities because of their large accumulations of woody biomass.

The utility of managing terrestrial ecosystems to conserve or sequester atmospheric carbon must be measured against the policy objectives for altering carbon emissions. Carbon emissions from the use of fossil fuels in the US were about 1,300 Mt-C per year (Mt = million metric ton = 10^{12} g) in 1988 and, from 1980 to 1988, emissions increased at an average rate of 6.4 Mt-C per year (NAS 1991), with the largest increase in the later years. Emission increases were greater toward the end of this period, reflecting increased economic activity. Thus, a policy objective of stabilizing carbon emissions would require emissions reductions, or implementation of strategies for conserving/sequestering carbon, amounting to an additional 6 to 10 Mt-C per year each year. A policy objective of reducing emissions by 10 to 20% would require emission reductions or increased carbon conservation/sequestration amounting to 130 to 260 Mt-C per year.

This document presents a model of the current and future carbon budget associated with the forest ecosystems of the conterminous US. The focus is on effects of economic and environmental policy changes for the period of 1990-2040. In this study, the concept of "forest ecosystem" has been expanded to "forest sector" by including biomass that has been physically removed (harvested) for human use. The potential effects of climate change have not been incorporated; however, current research is directed at addressing this question. The specific objectives of this report are to:

1. Develop a methodology for quantifying the current and future forest sector carbon budget for the US.

2. Apply the methodology to quantify the current status of carbon pools

and flux within the US forest sector.

3. Apply the methodology to several scenarios based on alternative policy options.

4. Identify gaps and deficiencies in the data and in the modeling approaches that require further effort.

The specific policy options considered in the report were assembled by the EPA Office of Policy, Planning, and Evaluation (OPPE) (Table 1). They function as a preliminary look at some of the types of options available. They do not represent the full spectrum of options that could be considered (NAS 1991, Rubin et al. 1992). Furthermore, they may not necessarily be the "best" options. The intent was to develop a tool that can be used to give broad technical guidance in evaluating various policy options for conserving or sequestering carbon in the forest sector.

The basic approach to modeling the current pools and net flux of carbon in US forests has been to develop a database which links inventory data on volume and area of different forest types by age class, with stand-level carbon budgets which quantify the pools of carbon associated with each age class within those forest types. Changes in carbon storage in response to various forest management policy scenarios (Table 1) were then evaluated by examining changes in forest inventories from 1990 to 2040, as well as changes in the post-harvest product and landfill pools. The HARVCARB model (Row and Phelps 1990) was used to estimate changes in the amount of carbon in forest products and in landfills from material harvested from 1990 to 2040.

Changes in forest inventory on private timberlands were estimated by linking an economic model (TAMM), which estimated tree harvest volume by region (Haynes and Adams 1985), to a forest inventory model (ATLAS), which tracked forest age-class distribution by forest type and productivity class (Mills and Kincaid 1992). TAMM insures that the economic consequences of implementing the various policy options were allowed to impact the carbon budget. The inventory at any point in time was run through a forest carbon model (FCM) to estimate total forest carbon. The change in total forest carbon over a 10-year interval, divided by 10, gave the average annual flux for that interval.

For public lands, there are no stand-alone inventory models. Furthermore, inventory assumptions are embedded in a variety of harvest planning models. It was thus necessary to estimate future carbon pools by using data from the planning efforts at the level of individual National Forests and other public agencies.

Over half of the total timberland carbon for the base year was in the mineral soil. Tree carbon, which included coarse roots, was the next largest component at 31%, followed by woody debris (11%), forest floor (6%), and understory (1%). Total carbon storage in living trees on timberland in the US was estimated at 11.1 Gt-C (Gt = billion metric tons = 10^{15} g). This quantity conforms closely with the estimate based on biomass factors derived from Cost et al. (1990) and Koch (1989), and to the estimates of Birdsey (1992a). The addition of woodlands carbon brought the total for all forests in the US to 11.6 Gt-C.

For the base year, the net effect of the biological and harvest-related transfers

was a carbon accumulation of 91 Mt on US timberland. The majority of the carbon accumulation was in the tree (52 Mt) and woody debris (36 Mt) pools. The forest floor showed a small gain (5 Mt) and the understory a small loss (-2 Mt).

Harvested removals amounted to 139 Mt. If the harvested carbon entered into permanent storage, the forest product pool would constitute an additional sink of that amount for the base year. However, determination of the net carbon transfer required consideration of carbon emissions associated with the decay or burning of these and previously produced forest products. Our estimated of the net change in the products pool was 36 Mt-C per year gain.

Annual carbon emissions from wildfire on US forests vary widely between years. Based on examination of USDA Forest Service statistics on fire frequency and extent over the last several decades (A. Auclair, Science & Policy Associates, personal communication) an estimate of 20 Tg-C was used in the present analysis. Approximately one-third of the total emissions are associated with woodlands, which tends to maintain a long-term carbon equilibrium on those lands. Direct emissions from burning of timberlands were thus estimated at 13 Tg-C.

Summation of the carbon flux processes described here suggests that the forest sector in the conterminous US was a sink for about 114 (91+36-13) Mt-C per year of carbon for the base year 1990. By way of comparison, carbon dioxide emissions (as carbon) from fossil fuel and cement manufacture in the 1990 was about 1,300 Mt-C per year. Thus the forest sector offset about 8% of US carbon emissions in 1990.

It is worth noting that, in principle, forest products which are used to produce energy may act to offset carbon emissions from fossil fuel combustion in the energy sector. This quantity has not been specified in the current analysis. However, the development of biomass energy schemes offers a potential opportunity for using the forest sector to reduce carbon emissions.

The scenarios ranged from the Current Forest Plans, which is the base scenario and reflects current best estimates of future harvest levels on public lands, to scenarios that called for reduced harvests on public lands, increased recycling, or increased afforestation. The 1989 Resource Planning Act (RPA) Assessment scenario was included because it represents the most recent well documented scenario produced by the USDA Forest Service. Differences between the RPA and the Current Forest Plans scenarios are a function of policies already in place.

The outcomes of the scenarios associated with the various forest policy options predict that US forests will continue to accumulate carbon during each decade of the 50-year period (Table 2). The rate of carbon accumulation will, however, decline over time. Each scenario predicts that the private forests, as simulated by the TAMM economics model coupled with the ATLAS timber inventory projection model, will change from accumulating carbon to losing carbon between 2010 and 2040. Contributing factors include rising harvest rates, shorter average rotation lengths, and the conversion of the forest land base to other uses such as urbanization. Less rigorous data and models were available to project public forest inventories. Both available and

reserved public timberlands appear to act as carbon sinks throughout the simulation period and more than balance carbon losses from private timberlands in later decades.

Though this trend was universal among the scenarios, there were important distinctions among them which contributed to differences in carbon pool totals and in temporal variations in carbon flux. The 1990 total carbon pool of US forests was estimated to be 36.2 Gt. Under the now obsolete 1989 RPA scenario, this figure was predicted to rise to 37.6 Gt in 2040. Increased carbon accumulation in public forests accounted for all of this difference. Although private timberlands had increased total carbon at the midpoint of the simulation, the total declined to the 1990 level by the simulation's end.

The Current Forest Plans marked a significant departure from the RPA scenario. A substantial portion of the public forest land base was shifted from the available category, i.e. forests subject to harvest, into long-term preserves. As a result, carbon accumulation in public forests increased, with carbon uptake rates 7 to 14 Mt-C per year higher than the RPA scenario over the course of the simulation. Harvest rates on the private forests increased slightly in response to the change in public forest management policies. From 1990 to 2010, carbon uptake was 2 Mt-C per year less on private lands than in the RPA scenario. However, this increased rate of harvest was not sustained and by the 2030's carbon uptake was 1 Mt-C per year in excess of the RPA scenario. For 2040, the total US forest carbon pool is estimated at 38.0 Gt. Again, the carbon total for private timberlands changed less than 0.1 Gt.

In this preliminary analysis, there was little effect of the additional harvest reduction (0.36 billion cubic feet per year reduction in 2000) on National Forests. The growth and standing inventory for the available timberland in National Forests over the 50-year period was nearly the same for the Current Forest Plans and Reduced National Forest Harvest scenarios. The bulk of the additional harvest reduction occurs in the Rocky Mountain Region, yet the projected inventory there was virtually unchanged. As with the Current Plans scenario, there was not a sustained compensatory harvest increase on private lands. Because the estimates of future growth and growing stock volume were based on reports from individual National Forests, the detailed analysis of age class distributions needed to explain the insensitivity of the growth and growing stock inventory to harvest reductions was unavailable. Any firm conclusion regarding effects of these additional harvest reductions must therefore await more detailed treatment of long term shifts in stand age class distribution.

Both the recycling scenarios and the afforestation scenarios further enhanced carbon accumulation over the 50-year simulation period. The afforestation scenarios had greater impact on the 2040 carbon pool total: values ranged from 38.3 to 38.7 Gt versus 38.3 to 38.4 for the recycling policies. However, the High Recycling scenario manifested the largest immediate increase in carbon uptake. For the period 1990 to 2010, carbon accumulation rates were an average of 13 Mt-C per year higher than for the Current Forest Plans. By comparison, the most effective afforestation policy over this same period, the $220 million per year Parks and Hardie scenario, achieved a 8 Mt-C per year increase. The major benefits of afforestation policies are

delayed until the latter half of the simulation, as the growth rates of the young forests begin to substantially increase. For the period 2010 to 2040, the average carbon accumulation rate of the $220 million Parks and Hardie scenario is 19 Mt-C per year greater than the Current Forest Plans uptake rate. The High Recycling scenario achieved only a 6 Mt-C per year advantage over Current Forest Plans over this same period.

In the afforestation scenarios considered here, the maximum increase in the forest land base was 5 x 10^6 ha (12.3 x 10^6 acres). This resulted in carbon sink increase of nearly 16 Mt-C per year over the 50-year simulation. Nationally, however, there are approximately 47 million ha (116 x 10^6 acres) of marginally productive, privately-owned crop and pasture land that is biologically capable of supporting tree growth (Parks 1992). This land presents an important opportunity for expanding the area of forest land in the US, and thereby increasing the rate of carbon sequestration.

The result of the first combination scenario was essentially the same as the Moderate Recycling scenario, in part due to the apparently negligible effect of lower public forest harvest rates on the forest inventory (combination scenarios assume public lands management as described in the Reduced National Forest Harvest). However, as noted, there is considerable uncertainty about the data for public forests. The effect of increasing exports in the second combination was to significantly reduce the benefits of implementing the Moderate Recycling scenario. The third combination, combining the Moderate Recycling assumption with doubled export levels and the $110 million per year Moulton and Richards scenario, yielded a benefit significantly greater than the unaltered afforestation scenario. The increased timber supply associated with paper recycling and afforestation was greater than the harvest increases associated with the doubling of timber exports. The combination scenarios emphasized the significant impact which otherwise unconsidered factors, in this case timber export trends, may have upon the simulations described in this report.

The analyses of carbon budgets using the TAMM/ATLAS/FCM modeling framework reported here do not include possible consequences of climate change. Projected climate changes could significantly alter terrestrial carbon sources or sinks. Of particular concern is the carbon source which would be created by extensive forest dieback at the warmer, drier end of each tree species' range. Even where particular forest types would still be maintained, effects of climate change may include significant changes in forest productivity and, hence, on the forest sector carbon budget.

The major research issues related to advancing the modeling framework developed in this analysis involve improving the forest inventory for public lands, incorporating potential effects of climate change on forest distribution and productivity, and integration of the US forest sector model with comparable models being developed in other countries. Satellite remote sensing will provide a means for developing a simplified but functional forest inventory across all land ownership classes. Coupling of vegetation/climate models, process-based ecosystem models, and the existing TAMM/ATLAS/FCM framework will permit projections of climate change effects on the forest sector carbon budget. The agreement on global

warming, recently signed by many counties at the UNCED Conference, will foster development of comparable country-level carbon budgets. Interaction with the research groups formulating these budgets, and with the research community using atmospheric CO_2 concentrations to model terrestrial carbon sources and sinks, will promote improved understanding of anthropogenic influences on the global carbon cycle.

Considered as a whole, there are significant opportunities for conservation and sequestration of carbon in the forestry sector of the US and globally (Sampson and Hair 1992). Obviously, forest sector policies alone will not compensate for the fossil carbon emission increases projected over the coming decades (Lashof and Tirpak 1990, IPPC 1992a). However, they can contribute to moderating the overall trend towards an increasing atmospheric CO_2 concentration.

Table 1. Summary of policy scenarios.

1989 RPA Assessment	Based on 1989 estimates of future forest timber supply, demand, and harvest. Harvest (growing stock volume) on National Forests increases from 2.16 billion cubic feet in 1986 to 2.17 in 2000 and to 2.53 billion cubic feet in 2040. Population increases from 242 million to 333 million, and GNP grows by 2 to 3% per year, from 1990 to 2040. Wastepaper recycling increases from 20.9% in 1986 to 21.0% in 2000 and 28.0 % in 2040.
Current Forest Plans (base scenario)	Like the RPA Assessment, except harvest on National Forests decreases from 2.16 billion cubic feet in 1986 to 1.72 billion cubic feet in 2000, and then increases to 2.01 billion cubic feet in 2040.
Reduced National Forest Harvest	Like the Current Forest Plans scenario, except harvest on National Forests, decreases from 2.16 billion cubic feet in 1986 to 1.36 billion cubic feet in 2000 and then increases to 1.59 billion cubic feet in 2040.
Moderate Recycling	Like the Current Forest Plans, except wastepaper recycling, increases from 20.9% in 1986 to 28.5% in 2000 and 41.5% in 2040.
High Recycling	Like the Current Forest Plans, except wastepaper recycling, increases from 20.9% in 1986 to 45.0% in 2000 and remains constant through 2040.
Afforestation Moulton and Richards $110 million	$110 million per year for 10 years is used in afforestation of pasturelands and marginal croplands (2.3 x 10^6 ha). Assumptions of annual costs and forest growth rates are from Moulton and Richards (1990). Otherwise follows Current Forest Plans.
Afforestation Moulton and Richards $220 million	$220 million per year for 10 years is used in afforestation of primarily pasturelands (4.0 x 10^6 ha). Assumptions of annual costs and forest growth rates are from Moulton and Richards (1990). Otherwise follows Current Forest Plans.
Afforestation Parks and Hardie $110 million	$110 million per year for 10 years is used in afforestation of primarily pasturelands (2.8 x 10^6 ha). Assumptions of annual costs and forest growth rates are from Parks and Hardie (1992). Otherwise follows Current Forest Plans.
Afforestation Parks and Hardie $220 million	$220 million per year for 10 years is used in afforestation of primarily pasturelands (5.0 x 10^6 ha). Assumptions of annual costs and forest growth rates are from Parks and Hardie (1992). Otherwise follows Current Forest Plans.
Combination 1	Includes the assumptions of the Reduced National Forest Harvest scenario and the Moderate Recycling scenario.
Combination 2	Includes the assumptions of the first combination scenario and an assumption of a doubling in timber exports.
Combination 3	Includes the assumptions of second combination scenario and the Low Moultan and Richards afforestation scenario.

Table 2. Summary of policy scenario results.

1989 RPA Assessment	Forest land accumulates carbon at a rate of 66 Mt-C per year in the 1990's after harvest removals are accounted for. The accumulation rate declines over time to 5 Mt-C per year because of a shrinking private forest land base, increased harvest levels and a shift to younger age classes.
Current Forest Plans (base scenario)	Public lands accumulate an additional 10 Mt-C per year, bringing the national sink to 74 Mt-C per year in the 1990's. The downward trend over time in the carbon accumulation rate is similar to the RPA Assessment scenario.
Reduced National Forest Harvest	Harvest reductions are likely to retain an additional several Mt-C per year on public lands.
Moderate Recycling	Carbon uptake, relative to Current Forest Plans, gradually increases to a peak of 13 Mt-C per year greater in 2020's. Carbon pool increase by 2040 is 75% of that achieved by High Recycling.
High Recycling	Reduced harvests on private lands lead to a carbon accumulation rate of 8 Mt-C per year greater than the Current Forest Plans in the 1990's, and 17 Mt per year greater in the 2000's. The difference declines in the later decades, and the same trend of reduction in the carbon accumulation rate over time is observed. Carbon pool in 2040 is 0.4 Gt greater than Current Forest Plans estimate.
Afforestation Moulton and Richards $110 million	Afforestation in 1990's enhances carbon accumulation relative to Current Forest Plans by 13 Mt-C per year in 2010's. Carbon pool value in 2040 is 0.4 Gt greater.
Afforestation Moulton and Richards $220 million	Carbon accumulation gains over Current Forest Plans are about 150% of the gain from the $110 million per year scenario.
Afforestation Parks and Hardie $110 million	Essentially the same result as the Moulton and Richards $110 million scenario.
Afforestation Parks and Hardie $220 million	Most effective of the afforestation scenarios. Carbon pool in 2040 is 0.8 Gt greater than Current Forest Plans.
Combination 1	Very similar to Moderate Recycling.
Combination 2	Reduces the carbon accumulation rates of Combination 1, in excess of Current Forest Plans, by 50% from 2010 to 2040.
Combination 3	Increases the carbon accumulation rate from Combination 2. Carbon pool in the year 2040 is 0.5 Gt greater than the Current Forest Plans scenario.

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SECTION 1. INTRODUCTION

A. Scope and objectives

Rising levels of atmospheric CO_2 and other radiatively important trace gases (e.g., CH_4, N_2O) from anthropogenic sources are likely to induce changes in the earth's climate over the coming decades (IPCC 1990). While there remains considerable uncertainty about the rate and magnitude of the possible climate change, there is an emerging consensus that policies relevant to stabilizing or reducing the emission of CO_2 and other "greenhouse gases" should be explored (NAS 1991, Rubin et al. 1992).

Terrestrial ecosystems have a major role in the global cycling of carbon (Figure 1.1). Vegetation and soil contain about three times as much carbon as the atmosphere. Also, each year, they exchange about 20 times as much carbon with the atmosphere as is emitted from fossil fuel use. For this reason, one proposed approach for slowing the increase in atmospheric CO_2 is to manage terrestrial ecosystems to conserve or sequester carbon (Schroeder and Ladd 1991, Sedjo and Solomon 1989, Sampson and Hair 1992) by changing the balance between the uptake of carbon dioxide from the atmosphere through photosynthesis and the release of carbon dioxide through respiration and biomass burning. Ecosystems in which release of CO_2 exceeds uptake are "net sources" of CO_2; if uptake from the atmosphere exceeds release, then they are "net sinks" of CO_2 from the atmosphere. In this report, "conservation" means that sources are being managed to decrease the net release of CO_2. Examples of conservation include decreasing deforestation rates or decreasing loss of soil carbon from disturbed systems. "Sequestration" refers to management of ecosystems to establish or augment sinks of CO_2. Reforestation, afforestation, and rehabilitation of degraded land are examples of sequestration.

Forest ecosystems are particularly important in consideration of carbon conservation and sequestration opportunities because of their large accumulation of woody biomass over time. Globally, forests contain about 60% of terrestrial carbon (Waring and Schlesinger 1985). In 1989, the Noordwijk Ministerial Conference recognized the importance of forests in global carbon cycling and established a provisional goal of increasing world forests at a rate of 12 million ha yr^{-1} (Noordwijk Conference Report 1989). Research on national-level forest carbon budgets is proceeding in several countries around the world, notably Canada (Apps and Kurz 1991), Russia (Kolchugina and Vinson, in press), Sweden (Eriksson 1991), and New Zealand (Hollinger and Hunt 1992). In Joensuu, Finland, in May 1992 a meeting, sponsored by the Intergovernmental Panel on Climate Change (IPCC), focused on appropriate approaches and methodologies for modeling the forest sector carbon budget (IPCC, in press). This meeting followed earlier IPCC meetings (Bangkok, Thailand 1991; Canberra, Australia 1992) on the potential role of forests in mitigating global climate change

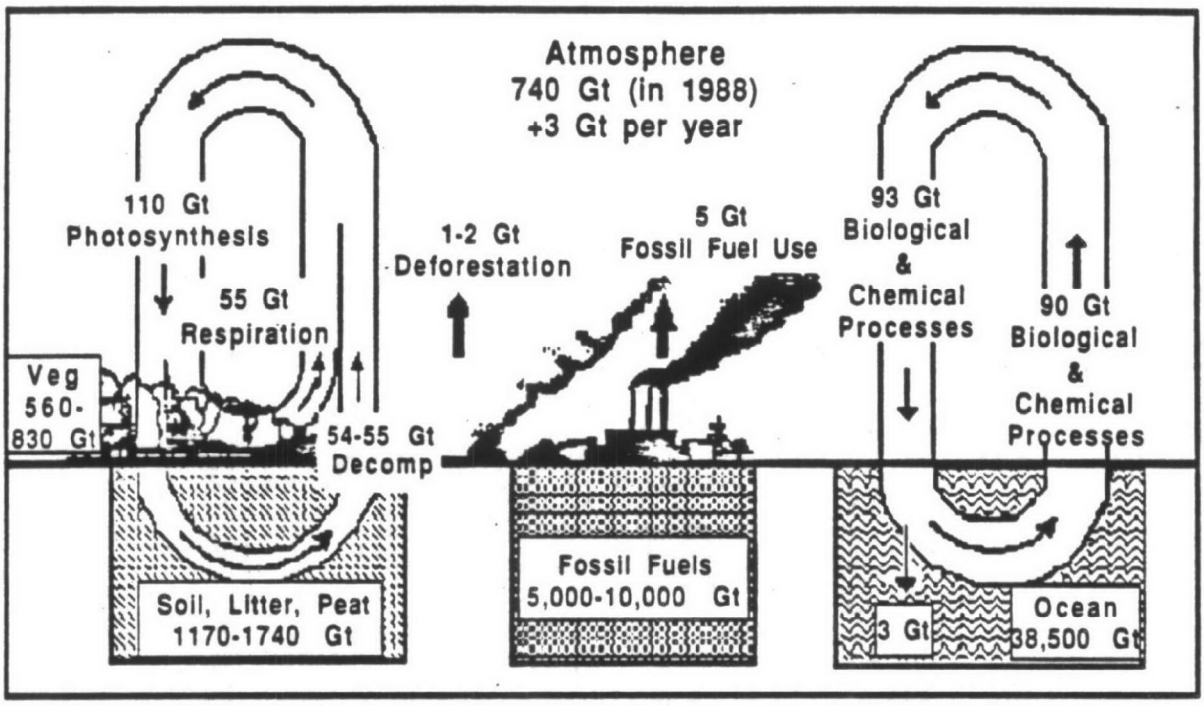

Figure 1.1. The global carbon cycle, including major pools and annual flux of carbon (Schneider 1989b).

(Howlett and Sargent 1991; IPCC 1992b). A global assessment of promising management practices concluded that conservation/ sequestration of carbon in forests was highly competitive with other options to mitigate greenhouse gas emissions (Dixon et al. 1991a).

The Climate Change Convention Framework promulgated by the US and other countries at the June 1992 United Nations Conference on Environment and Development (UNCED) calls for completion of comprehensive national-level carbon budgets for all countries. Biologically-based emission sources such as deforestation make a significant contribution to global CO_2 emissions. Biologically-based sinks, such as growing forests, may also have a global scale impact. Thus, forest land carbon budgets are an essential component of the national analyses. UNCED also enumerated several forestry principles which must be considered when evaluating options for managing forests to conserve/ sequester atmospheric carbon (Box 1).

The utility of managing terrestrial ecosystems to conserve or sequester atmospheric carbon must be measured against the policy objectives for altering carbon emissions. Currently, carbon emissions from the use of fossil fuels in the US are about 1,300 Tg-C yr^{-1} (Tg = 10^{12} g = 10^6 tons = megaton). From 1980 to 1988, emissions increased at an average rate of 6.4 Tg-C yr^{-1} (NAS 1991). Emission increases were greater toward the end of this period because of increased economic activity. Thus, a

Box 1. UNCED Forestry Principles

Proposals to manage forests to conserve and sequester atmospheric carbon must be considered in the context of many potentially conflicting or complimentary management options. Demographic and environmental pressures in many parts of the world are straining the capacity of forests to continue to supply basic goods and services (Mather 1990). As forests are exploited, serious impacts include loss of biodiversity, soil erosion, and reduced biomass productivity. These impacts and others affect the economic status of nearly 2 billion people worldwide whose livelihoods depend on forest systems. Multiple stresses resulting from global climate change projected for the next half century could further contribute to adverse forest impacts (Smith and Tirpak 1989).

In 1990, the G-7 nations (Canada, France, Germany, Great Britain, Italy, Japan, and the United States) agreed to support the concept of developing an international agreement on the use and management of forest resources (Maini 1991). The United Nations Conference on Environment and Development (UNCED), recently concluded in Rio de Janeiro, addressed this topic specifically. The result of the UNCED negotiations was a "non-legally binding authoritative statement of principles for a global consensus on the management, conservation, and sustainable development of all types of forests" (UNCED 1992).

The statement of principles contains 51 paragraphs covering a wide range of topics. Paragraphs addressing major themes affecting forest management include:

- Forests are related to economic development and any evaluation of forest policy should take place within the broader context of national development.
- Countries have sovereign rights to sustainably use their forest resources as needed. These rights include all biological resources, including genetic material.
- All population groups should be respected and included in the formulation of forest policy. This includes local populations, indigenous people, and women.
- International cooperation is critical for financing conservation, cost sharing, redressing external debt of developing countries, technical assistance, and technology transfer.
- Efforts should be undertaken toward greening the world, including reforestation, afforestation, and forest conservation.
- Research, institutional, and educational capabilities should be strengthened and there should be a free international exchange of research results.
- Free trade in forest products should be facilitated and trade barriers to higher value-added products should be reduced or removed.

policy objective of stabilizing carbon emissions would require emissions reductions, or implementation of strategies for conserving/sequestering carbon, amounting to an additional 6 to 10 Tg-C yr^{-1} each year. A policy objective of reducing emissions by 10 to 20% would require emission reductions or increased carbon conservation/ sequestration amounting to 130 to 260 Tg-C yr^{-1}.

This document presents a model of the current and future carbon budget associated with the forest ecosystems of the conterminous US. The focus is on effects of economic and environmental policy changes for the period of 1990-2040. In this study, the concept of "forest ecosystem" has been expanded to "forest sector" by including biomass that has been physically removed for human use (Box 2). The potential effects of climate change have not been incorporated; however, current research is directed at addressing this question. The specific objectives of this report are to:

1. Develop a methodology for quantifying the current and possible future forest sector carbon budget for the US.

2. Apply the methodology to quantify the current status of carbon pools and flux within the US forest sector.

3. Apply the methodology to several scenarios based on alternative policy options.

4. Identify gaps and deficiencies in the data and in the modeling approaches that require further effort.

Although the specific policy options considered in the report are realistic (Table 1.1), they do not represent the full spectrum of options that could be considered (NAS 1991; Rubin et al. 1992). Furthermore, they may not necessarily be the "best" options. The intent is to develop a tool that can be used to give broad technical guidance in evaluating various policy options for conserving or sequestering carbon in the forest sector.

B. Overview of modeling approach

The basic approach to modeling the current pools and net flux of carbon in US forests (see Figure 1.2) has been to develop a database which links inventory data on the areal extent of different forest types and age classes, with stand-level carbon budgets which quantify the pools of carbon associated with each age class within those forest types (Sections 3.A, 3.B). The inventory itself then provides the basis for an estimate of the carbon pool sizes and the age-specific increments; that is, differences between pool sizes in successive age classes provide the basis for a flux estimate. The forest inventory data and the stand-level carbon budgets described in this report are the primary inputs to the Forest Carbon Model (FCM) which calculates total carbon pools. Additional information on harvesting, wildfire, and change in the pool of forest products still in use and in landfills was used to estimate a base year carbon flux.

The inventory of forest carbon and the forest land base itself can change substantially as a result of economic and policy drivers (Box 3). In this study, changes in carbon storage in response to various forest sector policy scenarios (Table 1.1) were evaluated by examining changes in forest inventories from

Box 2. What is a Forest-sector Carbon Budget?

A carbon budget is a bookkeeping system for tracking the amount of carbon in various reservoirs ("pools"), and the amount of carbon transferred among the reservoirs and between the reservoirs and the atmosphere ("flux") (see figure below). The forest sector carbon budget, as referred to in this report, has three major pools: the organic matter of forest ecosystems, the products derived from forests, and products discarded into landfills. The major pools are, in turn, made up of subsidiary pools. For example, important pools within forests are trees, other vegetation, soil, the forest floor, and woody debris. "Net flux" is the difference between total uptake into a pool and total output from the pool, and is equal to the change in the pool size during some time interval.

The main uptake of carbon into forests is through photosynthesis, the fixation of atmospheric CO_2 by green plants ("primary producers"). Carbon loss from forests is mainly through respiration by green plants and other organisms, through burning, and through harvest of wood by humans. The first two processes result in the direct release of carbon to the atmosphere. The difference between photosynthesis and respiration by green plants is Net Primary Production (NPP). The difference between photosynthesis and respiration by all organisms, including decomposers (i.e., Net Ecosystem Production, NEP), is the total change in the carbon content of forests due to biological processes. Thus, NEP is also the net flux of carbon from the atmosphere to the forest from biological processes. The "net accumulation" by forests is NEP minus carbon removed by harvest. NPP, NEP, and the flux among the carbon pools within the forest (i.e., internal forest dynamics), are determined by stand characteristics such as tree species, age class, and site productivity. Regional and national estimates are obtained by combining forest stand NEP estimates with forest inventory data. In contrast, the harvest and reforestation rates are determined by external economic and policy factors. This report describes the methods and results of analyzing internal forest dynamics to estimate regional and national NEP, and the coupling of forest dynamics to the external economic and policy effects.

Unlike the forests, there are no systematic inventories of the product and landfill pools or of the net annual transfer into or out of these pools. The net annual change in carbon stored in the forest products pool can be estimated as the difference between published national values for the production of forest products and the transport of forest products to landfills. The national net change in the pool of forest products in landfills can be estimated as the difference between the amount of forest products transported to landfills and the carbon emissions to the atmosphere from landfills due to forest products estimated proportion of emission of carbon from landfills to the atmosphere due to forest products.

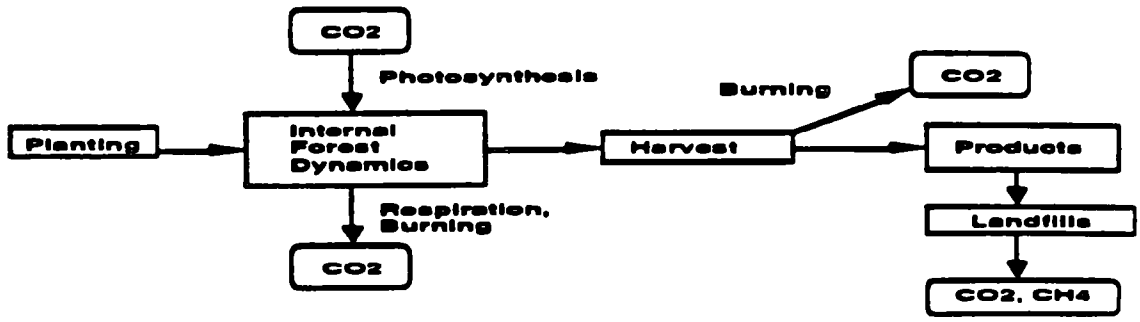

Table 1.1. Specifications for the policy options. See Table 1 (Executive Summary) and the text for additional characterization.

Scenarios	National Forests Harvest ($x\ 10^9\ ft^3$)		Recycling Intensity (%)		Afforestation ($x\ 10^6$ ha)
	2000	2040	2000	2040	
1989 RPA Assessment	2.17	2.53	21.0	28.0	---
Current Forest Plans (base year)	1.72	2.01	21.0	28.0	---
Reduced National Forest Harvest	1.36	1.59	21.0	28.0	---
Moderate Recycling	1.72	2.01	28.5	41.5	---
High Recycling	1.72	2.01	45.0	45.0	---
Afforestation Moulton and Richards $110 million	1.72	2.01	21.0	28.0	2.3
Afforestation Moulton and Richards $220 million	1.72	2.01	21.0	28.0	4.0
Afforestation Parks and Hardie $110 million	1.72	2.01	21.0	28.0	2.8
Afforestation Parks and Hardie $220 million	1.72	2.01	21.0	28.0	5.0
Combination 1	1.36	1.59	28.5	41.5	---
Combination 2*	1.36	1.59	28.5	41.5	---
Combination 3*	1.36	1.59	28.5	41.5	2.3

*Additional assumption of a doubling of timber exports.

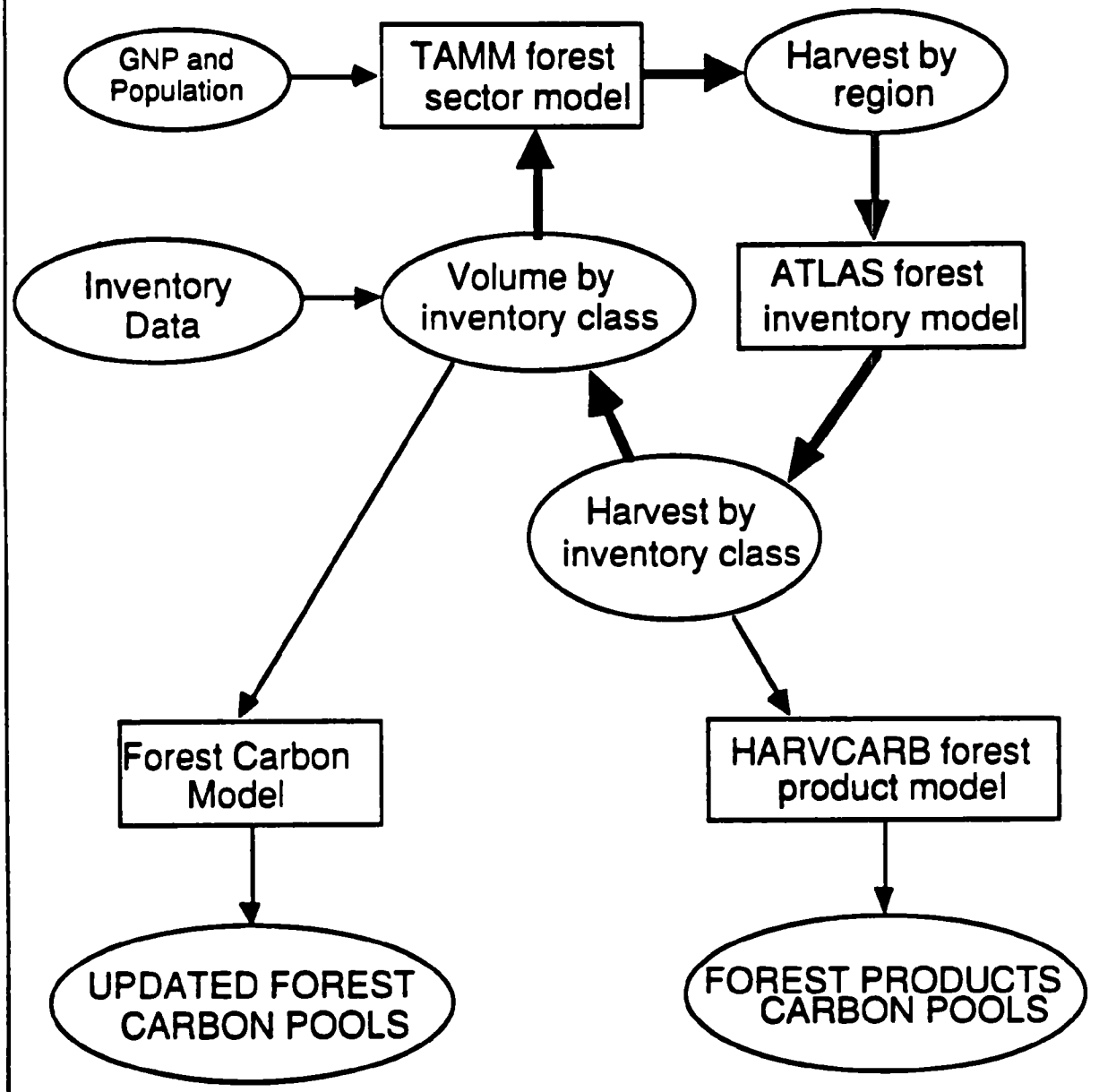

Figure 1.2. Modeling framework for forecasting carbon pools.

Box 3. Changes in the Size of the US Forest Land Base

About 25% of the area of the United States is occupied by closed forests, 226 million ha. This represents 8% of the world's closed forests. Approximately 45%, or 102 million ha, of US closed forests are managed. This is 10% of the world's managed forest area. Other wooded areas cover 71 million ha, or 8% of the country. The US accounts for 17% of annual global reforestation, i.e., replanting after harvest, about 1.8 million ha yr^{-1} (WRI 1992).

From the arrival of European settlers around 1630 until the 1920's, the area of commercial forest land in the United States decreased from 344 million ha to about 200 million ha (Clawson 1979, Williams 1988). This represents a decline of over 45% which was caused primarily by clearing for agriculture and by industrial felling and lumbering. Some recovery associated with abandonment of farmland and its reversion to forest, was evident through the 1960's but more recently timberland area has began to decrease again (Alig et al. 1990).

The future changes in forest land area in the US are highly dependent on economic and political factors. The study of Alig et al. (1990) suggested a slow decrease in the area of private timberland over the next 50 years. Alternatively, large areas of public and private land might be planted.

1990 to 2040, as well as changes in the post-harvest product and landfill pools, as projected by models of various types (Figure 1.2). Changes in forest inventory on private timberlands were estimated by linking an economic model (TAMM), which estimates tree harvest volume by region (Haynes and Adams 1985), to a forest inventory model (ATLAS), which tracks forest age-class distribution by forest type and productivity class (Mills and Kincaid 1992). TAMM insures that the economic consequences of implementing the various policy options are allowed to impact the carbon budget. The inventory at any point in time is run through FCM (Section 3.A) to estimate total forest carbon. The change in total forest carbon over a 10-year interval, divided by 10, gave the average annual flux for that interval. The model HARVCARB (Row and Phelps 1990) was used to estimate changes in the amount of carbon in forest products and in landfills from material harvested from 1990 to 2040.

For public lands, there are no stand-alone inventory models. Furthermore, inventory assumptions are embedded in a variety of harvest planning models. It was thus necessary to estimate future carbon pools by using data from the planning efforts at the level of individual National Forests and other public agencies.

C. Overview of this report

This document examines the role of biogeochemical processes and harvesting in the current forest land carbon budget of the US and evaluates several forest policy scenarios in terms of their impact on carbon conservation and sequestration. We begin with background material on the global carbon cycle, and on other efforts to model national forest sectors (Section 2). We then describe methods used in evaluating (1) the current status of carbon

stored in various parts (i.e., "pools") in forests and other terrestrial ecosystems in the US, (2) the amounts of carbon exchanged with the atmosphere or removed through harvests each year (i.e., net flux), and (3) the projected changes in carbon pools and flux through the year 2040 as to several policy options that might alter these pools and flux (Section 3). This is done in the context of anticipated economic demand for forest products for the period 1990-2040.

Results of these analyses are presented in Section 4 along with a discussion of the policy considerations. Section 5 introduces the potential effects that climate change might have on forest carbon pools and flux, and Section 6 discusses research needed to begin assessing the effects of climate change on national carbon budgets. Other research needed to refine and extend the Forest Carbon Model is also discussed. A set of supporting documents is presented in Section 9. These include a discussion of uncertainties in the inventory data, an estimate of the carbon in woodlands of the US, a documentation of the approach used in estimating woody debris pools, an estimate of the carbon flux associated with nonforest lands, and a glossary.

SECTION 2. BACKGROUND: THE TERRESTRIAL CARBON BUDGET

A. Introduction

The CO_2 concentration in the atmosphere is increasing at a rate of approximately 1.5 ppm per year as a result of fossil fuel combustion and deforestation (IPCC 1990). As atmospheric CO_2 levels rise, global temperatures may increase through the greenhouse effect (Hansen et al. 1988, Schlesinger and Mitchell 1985). Predictions are that average global temperatures may increase by 1.5 to 4.5^{o} C in the first half of the next century.

The potential consequences of increasing atmospheric carbon levels are important enough that governments are considering programs to stabilize/reduce anthropogenic emissions and increase forest-carbon sequestration (Lashof and Tirpak 1990, Downing and Cataldo 1992). However, to develop effective policy options, an understanding of the global carbon cycle, and how anthropogenic carbon emissions may affect it, is necessary.

Our understanding of the global carbon cycle (i.e., sources and sinks) suggests that managing terrestrial ecosystems, especially forests, to sequester and conserve carbon may contribute to moderating the rise in atmospheric CO_2 (Box 4; Sampson and Hair 1992). However, better knowledge of carbon pools and transfer rates is needed. Such information will be crucial in formulating national and international policy to reduce the risks of altering the composition of the atmosphere.

B. The global carbon cycle

The global carbon cycle is the movement of carbon among the atmosphere, terrestrial biosphere, and ocean pools (Schneider 1989a, Post et al. 1990, Dixon and Turner 1991, Simpson and Botkin 1992). Although the general nature of the global carbon cycle is well known, relatively large uncertainties exist in the estimation of the magnitude of the global carbon pools and the flux among them. On a global scale, the amount of carbon in the different pools has been estimated by a variety of means (Whittaker and Likens 1975, Ajtay et al. 1979, Bolin et al. 1979, Olson et al. 1983, Schlesinger 1984, Mooney et al. 1987, Schneider 1989, Schneider 1989a, Downing and Cataldo 1992, Simpson and Botkin 1992). Current estimates suggest that oceans store by far the largest amount of carbon at 38,500 Gt-C (Gt = 10^9 tons = Gigaton = 10^{15} g). The second largest reservoir is fossil fuels with estimates ranging from 5,000 to 10,000 Pg-C. Soil is the next largest pool at 1170 to 1740 Pg-C. The vegetation and atmospheric carbon reservoirs are about equal and are the smallest in comparison to the others. Estimates of carbon storage in vegetation ranges from 560 to 830 Pg-C. The atmospheric pool contains approximately 740 Pg-C of carbon.

Even though the terrestrial biosphere, vegetation and soils, store much less carbon than does the oceans, the amount of carbon that cycles annually

Box 4. Global View of Carbon Conservation/Sequestration

In 1991, a global assessment was undertaken of the potential of forest management practices to store atmospheric carbon (Dixon et al. 1991a; Winjum et al. 1992; Dixon et al., in press; Winjum et al., in press; Schroeder et al., in press). The assessment was based on information on the rates of carbon storage per hectare for many practices, their implementation costs, and estimates of the amounts of land suitable for forest management. Information was compiled through a survey of current published technical literature for forested nations representing boreal, temperate and tropical regions of the world. Key findings of the assessment are highlighted in the following paragraphs. Because of the geographic scope of the 1991 assessment, it was necessary to use generalized representations of ecoregions and data. Thus, the results are broadly applicable for comparing ecoregions and countries. However, they lack the spatially detailed basis needed for within-country analyses such as that presented for the US in this document. Also, the data for the 1991 assessment did not consider change over time so that it was not possible to estimate carbon flux.

Carbon Storage

The assessment indicated that the most promising forest management practices to sequester carbon in the terrestrial biosphere include: reforestation in the temperate and tropical latitudes, afforestation in the temperate regions, and agroforestry and natural reforestation in the tropics. Least promising from a carbon storage standpoint were the application of silvicultural practices, such as thinning, fertilization and other stand improvement treatments, at all latitudes.

The potential carbon storage ranges of forestation and silvicultural practices by major latitudinal biomes were:

	Forestation	Silviculture
	----------t C/ha----------	
Boreal	15-40	3-10
Temperate	30-180	10-45
Tropical	30-130	14-70

Costs of Storing Carbon

The median cost efficiency for all management practices in terms of establishment costs was about $5/t-C with an interquartile range (middle 50% of observations) of $1 to $19/t-C. The most cost-efficient forestry and agroforestry practices, based on establishment costs, within zones of latitude are shown in the table below.

Total costs per ton of carbon sequestered, which include land rental, would be considerably higher. The magnitude of the total costs is difficult to estimate because of economic uncertainties regarding factors such as land rental.

Suitable Land Area

Preliminary estimates indicated that there are large amounts of land which are technically suitable for expanding the area of managed forest and agroforestry systems. Approximate areas of suitable land by latitudinal regions appear to be: boreal, 100 to 200 million ha (Volz et al. 1991); temperate, 200 to 300 million ha (Moulton and Richards 1990, Volz et al. 1991); and tropics, 420 million (Grainger 1991) to almost 2 billion ha (Houghton et al. 1991). These estimates of technically-suitable lands must be viewed with caution. The amount of land actually available from

a social and economic standpoint remains uncertain, but it is likely to be much less, in most cases, than the land area that is technically suitable (Trexler 1991a). The fallow stage of a shifting agricultural cycle serves as an illustration. Technically, such land is suitable for natural vegetative regeneration and the accompanying accumulation of biomass and carbon. In reality, such land may be a part of an important mosaic of land use patterns that is required to support the agricultural requirements of local populations. Therefore, more complete data on the amount of land actually available for expanding forest and agroforestry systems are clearly needed, and research efforts are underway to achieve this objective.

Establishment costs of cost-efficient practices.

	Median $/tC	Median $/ha
Boreal:		
Natural Regeneration	5 (4-11)	93 (83-126)
Reforestation	8 (3-27)	324 (127-455)
Temperate:		
Natural Regeneration	1 (<1-1)	9 (9-10)
Afforestation	2 (<1-5)	259 (41-444)
Reforestation	6 (3-29)	357 (257-911)
Tropical:		
Natural Regeneration	1 (<1-2)	178 (106-238)
Agroforestry	5 (2-11)	454 (255-699)
Reforestation	7 (3-26)	450 (303-1183)

*Interquartile ranges are shown in parentheses

between the terrestrial biosphere and the atmosphere is about equal to that which cycles between the oceans and the atmosphere. Their combined annual flux is about 30% of the total atmospheric carbon pool.

A key concern, from the perspective of possible global climate change, is that the flux of CO_2 in and out of the atmosphere is not balanced. That is, more carbon, mainly in the form of CO_2, is accumulating in the atmosphere than is being removed through plant photosynthesis and abiotic processes. It is estimated that the atmosphere is gaining about 3 Pg of carbon per year (Schneider 1989a, Tans et al. 1990). The two principal sources of the atmospheric CO_2 are fossil fuels combustion (Marland 1989, Schneider 1989b) and deforestation (Houghton et al. 1983, Woodwell et al 1983, Houghton et al 1985, Palm et al. 1986, Detwiler and Hall 1988). The annual contributions of these two anthropogenic carbon sources total 6 to 8 Pg with 1-2 Pg derived from deforestation. Atmospheric CO_2 contributions from deforestation and fossil fuel combustion are likely to increase (IPCC 1990).

One of the critical tasks in assessing recent changes in the carbon cycle is determining where the anthropogenic carbon emissions are accumulating, since only about half remains in the atmospheric pool. The two possibilities are the oceans and the terrestrial biosphere. Some recent analyses suggest that temperate and tropical forest ecosystems are sequestering more carbon than previously reported (Sedjo 1992, Sedjo and Solomon 1989). However, large uncertainties remain and although the potential for unmanaged and managed forest ecosystems to sequester and conserve atmospheric carbon appears to be large the current sequestration rate and the potential for additional sequestration needs to be better quantified.

A recent workshop on the global carbon cycle evaluated the role of terrestrial carbon sinks (Lugo and Wisniewski 1992). Some of the major conclusions and recommendations of the workshop were:

- The global carbon cycle is important but poorly understood.
- The exchange of CO_2 between biota and the atmosphere has not been in steady state for several centuries. Many terrestrial ecosystems are currently accumulating carbon.
- Natural plant succession after disturbance (e.g., deforestation, forest tree die back) may sequester large amounts of atmospheric carbon. It is important to understand the role that the various seral stages may serve as sources or sinks of carbon.
- Increasing levels of atmospheric CO_2 may increase plant productivity and may increase carbon storage in terrestrial ecosystems.
- The carbon cycle is inextricably linked to nutrient and hydrological cycles.
- Proactive terrestrial ecosystem management can enhance carbon sequestration and conservation and is compatible with biodiversity conservation, sustainable land use, and economic development.

C. Regional carbon budgets

The global carbon budget quantifies the carbon cycle by describing the transfer

of carbon among the various reservoirs (Bolin et al 1979, Post et al. 1990). In recent attempts to quantify the global carbon cycle a discrepancy has arisen between results of models which focus on the time course in the change in the atmospheric CO_2 concentration and models which reconstruct the time course of deforestation and land use change.

Reconciliation of these modeling approaches will require critical analysis of assumptions about long-term changes in terrestrial sources and sinks of carbon (Lugo 1992). Attention to possibly oversimplified approaches for representing terrestrial systems is also needed (Lugo 1992, Downing and Cataldo 1992). For example, a recent IPCC report (Melillo et al. 1990) used only two biomes to represent all of the tropics. Such reductive models lump too much variety into a few biome categories. This results in poor estimates of carbon pools and flux because sources and sinks are averaged without proper weighting.

Lugo and Brown (1991) argued that each tropical forest life zone has its own unique biota, history of land-use management, and set of carbon fluxes and sinks, and therefore, should be evaluated separately to quantify carbon pools and transfer rates. This same reasoning should also extend to temperate and boreal forests and to other biomes (e.g., grassland, desert) as well.

These considerations point towards the need for more refined carbon budgets at the regional or national scale. Circumboreal countries such as Finland have characteristic sinks for carbon in the form of peatlands (Laine and Päivänen 1992). More highly forested countries may have greatly differing rates of tree growth, harvest and disturbance, all of which strongly impact carbon flux. Data on rates of deforestation, reforestation, forest stand classification and condition, and fossil-fuel combustion are usually specific to a region/country and much more accurately determined on geo-referenced basis than using a global approach.

Region- or country-level carbon budget analyses are important since they can add to or show the inconsistencies within the global-scale analyses (Sedjo 1992). Such geographically specific data bases may verify the model-based global carbon budgets, or they can raise questions as to their credibility and provide direction for additional research. Also, country-level studies will provide scientific data to help deliberations on policy issues regarding CO_2 emissions, deforestation, reforestation, and pending global climate change.

Some examples of region- and country-level, forest-sector carbon budget analyses are the tropical forests (Makundi and Sathaye 1992), temperate forests (Sedjo 1992), European forests (Kauppi et al. 1992), Canada (Kurz et al. 1992), Finland (Karjalainen and Kellomäki, in press), Sweden (Eriksson 1991), Netherlands (Heil et al., in press), Poland (Galinski 1992), the former Soviet Union (Kolchugina and Vinson 1992), and New Zealand (Maclaren and Wakelin 1991).

The value of these country level analyses can be illustrated by evaluating the carbon budget for New Zealand. The amount of carbon currently in the forests of New Zealand is approximately 1.13 Pg-C. An additional 0.22 Pg-C are stored in forest products such as building lumber and paper. New Zealand forests currently sequester approximately 3.5 Tg-C yr^{-1}. Carbon

dioxide releases into the atmosphere are estimated to be 6.0 Tg yr^{-1}. Obviously, New Zealand is a net source for CO_2. To achieve a favorable CO_2 balance, 50,000 ha yr^{-1} of reforestation is necessary to offset the current emissions. Reforestation at this rate would increase New Zealand's forest estate to 4.5 million ha by 2050, nearly 4 times its present size. Such information can be used by the New Zealand government in developing policy issues with regards to CO_2 emissions and reforestation.

D. Vegetation and the global carbon cycle

Terrestrial vegetation is an important component of the carbon cycle (Goreou 1990, Simpson and Botkin 1992). Plant photosynthesis removes CO_2 from the atmosphere while plant and microbial respiration returns CO_2 to the atmosphere. The carbon removed from the atmosphere through photosynthesis sustains plant growth and development. Carbon is then transferred to other organisms through a complex food web. The impact of terrestrial vegetation on atmospheric CO_2 levels is illustrated by its seasonal fluctuation (Bolin et al 1979, King et al. 1987, Keeling et al. 1989). In the northern hemisphere, atmospheric CO_2 is lowest during the summer growing season when photosynthesis is at a maximum, and highest during winter months when photosynthesis is minimal. Bolin et al. (1979) have estimated a 10% seasonal fluctuation in atmospheric CO_2 levels.

In addition to seasonal variations, vegetation can influence long-term atmospheric CO_2 levels through carbon sequestration (Simpson and Botkin 1992). Not all forms of vegetation have an equal potential to affect atmospheric CO_2. For example, trees and shrubs can store carbon for hundreds of years in woody plant parts. The carbon stored is only returned to the atmosphere through decomposition or burning. On the other hand, the carbon stored in non-woody plant parts and herbaceous plants is often returned to the atmosphere within a year of sequestration. Thus, over broad spatial scales vegetation can be either a carbon source or sink depending on many factors such as past land-use history, degree of disturbance, successional stage, stand age, management practices, and vegetation type (Detwiter and Hall 1988).

Deforestation and biomass burning are other mechanisms by which vegetation affects the global carbon cycle. Estimates of net global CO_2 emission from land-use change such as deforestation for 1980 are approximately 10 to 30% of annual anthropogenic emissions (Lashof and Tirpak 1990).

Vegetation also influences the carbon cycle indirectly through its effects on climate (Simpson and Botkin 1992). Energy and water balance simulations of tropical rain forests suggest that deforestation results in elevated albedo, decreased evapotranspiration, and increased soil temperatures; large-scale deforestation may affect regional precipitational patterns (Dickinson 1991). Such changes in climate can reduce the potential for plant growth and the rate of carbon sequestration.

The strong influence of vegetation on the global carbon cycle suggests that accurate assessments on the extent of major ecosystems and vegetation biomass densities with changes over time are needed (Simpson and Botkin 1992).

E. Mitigation of rising carbon dioxide levels

Atmospheric CO_2 has been increasing since the industrial revolution in response to anthropogenic emissions and deforestation (Keeling et al. 1989). Various policy options to moderate the rising CO_2 concentration have been presented that, if put into action, would decrease the risks of global climate change (Lashof and Tirpak 1990). Maintaining CO_2 emissions at current rates will not immediately reverse the trend of increasing atmospheric levels. Even if emissions remained at 1985 rates, the greenhouse effect would continue to intensify with CO_2 reaching 500 ppm by 2100 (Lashof and Tirpak 1990). A 50 to 80% reduction in CO_2 emissions would be necessary to stabilize atmospheric CO_2 at the current level of about 355 ppm. This is because CO_2 can have a long atmospheric residency time and maintaining current emissions rates will still result in rising levels. Approximately 50 to 100 years would be required following a drastic reduction in CO_2 emissions for the oceans and terrestrial biosphere sinks to substantially reduce atmospheric CO_2.

However, action can be taken to help moderate the current trend of rising atmospheric CO_2 and reduce the risks of global climate change (Lashof and Tirpak 1990). Action to reduce CO_2 emissions could include increased automobile fuel efficiency, increased use of alternative motor fuels or electricity, greater use of solar and nuclear energy, improved efficiency in industrial and residential energy use, and decreased deforestation.

Proactive reforestation and afforestation programs could also help to counterbalance the rise in atmospheric CO_2 levels (Dixon et al. 1991a). The goal of reforestation and afforestation programs would be to improve the carbon sink potential of terrestrial ecosystems. For this to occur, vast areas of suitable land need to be reforested. In the policy scenario analyses by Lashof and Tirpak (1990), reforestation was considered a significant policy option in decreasing the global climate change potential.

F. Forest management concerns, constraints, and approaches

Although forests occupy approximately 36% of the world's land surface, only about 10% receive any type of management (Allan and Lanly 1991, Goodland et al. 1990, Poore et al. 1990, WRI 1990). Obviously, natural resources management could play a much more important role in the global forest sector. In addition, the expansion or intensification of management practices could improve the goods and services obtained from global forests. One critical service derived from the forest sector, that needs more management focus within the context of pending global climate change, is the ability of trees to sequester and conserve atmospheric carbon (McG Tegart 1992).

Management strategies that would have immediate benefits in mitigating the atmospheric CO_2 concentration are those that promote a decreased rate of global deforestation and an increased of afforestation or reforestation of degraded lands capable of sustaining forest tree growth (Dixon and Andrasko 1992). Global deforestation has resulted not only in a reduction of forested land, but has also decreased plant and animal biodiversity, increased desertification including loss of plant cover and increased soil erosion, and released vast amounts of CO_2 into the

atmosphere (Maini 1991, NAS 1991). Given the overall decline in forested areas resulting from tree harvesting and land use change, interest in technically managing forests for sustainability of goods and services has gained momentum among natural resource managers and policy makers. Slowing the rate of deforestation and expanding reforestation projects throughout the world have been proposed as cost-effective means to sequester carbon, protect soils and watersheds, and provide a continual source of chemicals, food, fiber and fuel (Dixon et al. 1991b, Lugo and Wisniewski 1992).

For sustainable forest management to be successful in mitigating the buildup of atmospheric CO_2 through carbon sequestration, it is essential that it meet the social and economic needs of the local people (Gregerson et al. 1989). Forests, especially those located in tropical regions, have long been of economic importance and thus influenced greatly by anthropogenic use and pressure (Gregerson et al. 1989, Schneider 1989). Millions of hectares of forest land around the world are impacted annually by various activities such as lumber production, livestock grazing, or fuel wood harvesting (WRI 1990). However, intact, undisturbed forests may provide more value to the local economy in terms of food, fiber, chemicals, and biodiversity than degraded forest land (Peters et al. 1989). Economics may provide the strongest argument for sustainable forest management practices, at least at the local level, where benefits are directly realized by the community. This approach of sustainable forest management could lessen the pressure for further global deforestation (Sanchez 1990).

Forest scientists have proposed various strategies for using forest management techniques in adaptation to and mitigation of potential climate change (Andrasko 1990a, McG Tegart 1992, Trexler 1991b). Examples of such recommendations are as follows:

- Create financial and policy incentives to reduce rates of forest conversion for unsustainable agriculture, grazing, and forest harvesting.
- Reduce the frequency, interval, scale, and amount of forest and savannah consumed by biomass burning which is intended to create or maintain pasture and grassland.
- Introduce forest management systems which utilize low impact harvesting methods, replacing destructive high impact logging practices.
- Conserve standing primary and old-growth forests as stocks of biomass, offering a stream of economic benefits.
- Substitute technically sustainable technologies for slash-and-burn agriculture that requires deforestation.
- Decrease consumption of forests and trees for cash crops and development projects through environmental planning and management.
- Promote more complete utilization of forest products.
- Promote the recycling of forest products.
- Discourage the production of disposable forest products by substituting durable wood or other goods.
- Prevent loss of soil carbon by slow-

ing erosion in forest systems caused by harvesting and livestock grazing.

- Improve forest productivity on existing land through management and genetic manipulation.
- Establish plantations on surplus cropland and urban lands.
- Restore degraded forest and savannah ecosystems through natural regeneration, reforestation, and protection.
- Promote opportunities to increase soil carbon storage by leaving slash from tree harvesting.
- Improve the efficiency of biomass (fuel wood) combustion in cooking and heating stoves and industrial uses.

G. Conclusions

The general nature of the global carbon cycle is known. However, uncertainties exist in estimating carbon pools and the transfers among them in relationship with anthropogenic CO_2 emissions. The region- or country-level approach to identify carbon sources and sinks will aid in validating the global-scale analysis and provide the basis for policy decisions regarding anthropogenic carbon emissions and carbon sequestration. Until the time when the relationships among anthropogenic CO_2 emissions and biological carbon sinks are understood, there will be unresolved scientific issues that will hinder the policy making process.

SECTION 3. METHODS

A. Base year carbon pools

A comprehensive estimation of the magnitude of carbon pools on US timberland (Box 5) was conducted using yield tables associated with the Aggregate TimberLand Analysis System (ATLAS) and existing forest inventory databases. ATLAS is a timber inventory model developed for national timber resource assessment by the USDA Forest Service (Section 3.B, Mills and Kincaid 1992). ATLAS considers only growing stock volume. The yield tables in ATLAS were used as the basis for constructing stand-level carbon budgets which treated all major carbon pools. Carbon pool size per unit area per age class was thus estimated for the living tree biomass, woody debris, understory vegetation, forest floor, and soil components of the forest.

Separate stand-level carbon budgets were prepared for each of 422 yield tables supplied by the USDA Forest Service as input to ATLAS. Carbon storage at the regional and national level could then be determined by coupling these stand-level carbon budgets to forest inventory data on the areal extent and stocking volume of each age class within each inventory type. The set of stand-level carbon budgets provides the foundation for what is referred to in this document as the Forest Carbon Model (FCM).

1. Construction of stand-level carbon budgets

Living tree biomass

The construction of a stand-level carbon budget began with an ATLAS yield table which related stand age to the volume of growing stock (i.e., merchantable wood) at normal stocking levels. Classification of the ATLAS tables is by region, forest type, site productivity, ownership and management scheme. Age class increments in the tables are 5 or 10 years, depending on the region. The maximum stand age may reach 90 to 175 years, depending on region, forest type, and management practices.

Since the ATLAS yield tables include only growing stock volume, an adjustment factor was applied to account for noncommercial species (1.01 for softwoods, 1.14 for hardwoods; see Section 9.C). Noncommercial species include those of "small size, poor form, or inferior quality which normally do not develop into trees suitable for industrial wood products" (Waddell et al. 1989). Adjusted merchantable bole volume was converted to bole carbon based on the relative proportion of hardwood and softwood volume by age class, and forest-type-specific volume to carbon ratios for softwoods and hardwoods (Table 3.1, Birdsey 1992c). The subsequent conversion from merchantable bole carbon to total tree carbon was based on the analysis of Harmon (Section 9.C). His analysis used harvest statistics and allometry to account for the contribution of roots, stumps, branches, and tops (Column 2, Table 9.C.3).

Sapling carbon (trees less than 12.5 cm dbh) was included in the tree carbon pool estimates reported here. Carbon contributed by saplings was derived

Box 5. Definition of Timberland and Woodland

A primary functional classification of forest land in the US differentiates between timberland, reserved timberland, and other forest. Waddell et al. (1989) define timberland as:

Forest land that is producing or is capable of producing crops of industrial wood and not withdrawn from timber utilization by statute or administrative regulation. (Note: Areas qualifying as timberland are capable of producing in excess of 20 cubic feet per acre per year [1.4 m^3 ha^{-1} yr^{-1}] *of industrial wood in natural stands. Currently inaccessible and inoperable areas are included.)*

Reserved timberland refers to land having the productive capacity of timberland but which is withdrawn from timber utilization by statute or administrative regulation.

Other forest land includes:

Forest land other than timberland and reserved timberland. It includes available and reserved unproductive forest land, which is incapable of producing annually 20 cubic feet per acre per year of industrial wood under natural conditions because of adverse site conditions such as sterile soils, dry climate, poor drainage, high elevation, steepness, or rockiness.

Throughout this document, the term "woodland" has been used to refer to other forest land.

The analyses reported in this document are mostly confined to timberland forest, including reserved timberland. These lands account for the vast majority of the forest biomass and forest productivity of the conterminous US. Woodland acreage and biomass values are described in Box 6.

Table 3.1. Factors to convert tree volume (cubic feet) to carbon (kg). Conversion factors from Birdsey (1992a).

		Specific Gravity [1] g (wood)/cc (wood)		Percent Carbon [2] g-C/g (wood)		Factor [3] kg-C/ft^3 (wood)	
Region	Forest Type	Softwood	Hardwood	Softwood	Hardwood	Softwood	Hardwood
Southeast and South Central	Pines	.510	.639	.531	.497	7.68	9.01
	Oak-hickory	.536	.639	.531	.479	8.07	9.01
	Oak-pine	.523	.639	.531	.497	7.88	9.01
	Bottomland hardwoods	.460	.580	.531	.497	6.93	8.18
Northeast and Mid Atlantic	Pines	.378	.543	.521	.498	5.59	7.67
	Spruce-Fir	.369	.525	.521	.498	5.45	7.41
	Oak-hickory	.374	.636	.521	.498	5.53	8.98
	Maple-beech-birch	.384	.600	.521	.498	5.67	8.48
	Bottomland hardwoods	.460	.580	.521	.498	6.80	8 18
North Central and Central	Pines	.421	.530	.521	.498	6.22	7.49
	Spruce-Fir	.351	.480	.521	.498	5.19	6.78
	Oak-hickory	.416	.632	.521	.498	6.15	8.93
	Maple-beech	.372	.576	.521	.498	5.50	8 14
	Aspen-birch	.370	.465	.521	.498	5.47	6.57
	Bottomland hardwoods	.460	.580	.521	.498	6.80	8 18
Rocky Mountain and Pacific Coast	Douglas fir	.473	.380	.512	.496	6.87	5.35
	Ponderosa pine	.416	.380	.512	.496	6 04	5 35
	Fir-spruce	.349	.380	.512	.496	4.45	4 85
	Hemlock-sitka sp.	.434	.433	512	.496	5.53	5.53
	Lodgepole pine	.423	.380	.512	.496	5.39	4 85
	Larch	.508	.433	.512	.496	6.48	5.53
	Redwoods	.416	.580	.512	.496	5.31	7.40
	Hardwoods	.424	.384	.512	.496	5.41	4.89

1 Weighted average specific gravity of the three most common (in terms of volume) softwood or hardwood species within the forest type.

2 From Koch (1989).

3 Factor = specific gravity times the volume of a cubic foot (28 36 x 10^4 cc) times percent carbon

from the ratio of bole carbon to sapling carbon. These factors, specific to individual forest types, were based on regional relationships described by Cost et al. (1990) and were dependent on the predominant stand fiber type.

FCM treats the ratio for total tree carbon to merchantable bole carbon as a constant by forest type; in fact, the ratio varies with stand age. The ratio is highest early in stand development. Quantitative analyses in forest plantations suggest that after canopy closure the ratio is relatively stable. As a result, FCM underestimates total tree carbon in young stands. But because the absolute volume in young forests is low, the net effect of not accounting for the change in this ratio is small. In old stands, use of the fixed ratio would tend to overestimate tree biomass because the mass of branches tends to remain constant while the stem volume continues to increase. However, checks of biomass estimates for older age classes against biomass studies reported in the literature (e.g., Cannell 1982) did not reveal a strong bias. In future development of the FCM, this ratio and the wood density may be allowed to vary with stand age.

Woody debris

The woody debris pool consists of standing dead trunks, dead coarse (>2 mm diameter) roots, and dead woody material of greater than 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 (FIA) studies, we used an approach based on an analysis by Harmon (Section 9.C) to estimate woody debris pool sizes. The woody debris pool for each age-class was modeled as the woody debris pool at age 0 plus an increment due to mortality (Table 9.C.2), and minus a decrement due to decay (Table 9.C.4). The woody debris pool at age 0 was estimated differently for western and eastern forests. Western forests have undergone relatively few rotations; thus, we assumed that the woody debris from the primary forest was still present. The woody debris pool at age 0 was estimated as the steady state pool for a mature forest (Table 9.C.5), plus an increment due to residue from live trees left on-site after harvest (Table 9.C.3.).

Because many eastern forests have been harvested several times, we assumed that much of the debris from the primary forest has decayed. Woody debris at age 0 was estimated by an iterative process. The steady state value for mature forest (Table 9.C.5) was used as an initial "seed". The debris values for successive age classes were estimated by incrementing and decrementing this value for mortality and decay, assuming current stocking levels. The value at age 0 was then recalculated as the estimated debris value at rotation age plus an addition due to harvest residue. This procedure was iterated until woody debris at age 0 converged to a stable value.

Understory carbon

The understory carbon pool consists of herbs and shrubs. The pool sizes were estimated on a forest-type-specific basis and were based on relatively few examples in the literature (Plantinga and Birdsey, in press). The pool size rises rapidly after harvest or disturbance but falls as the canopy closes. It eventually begins rising again as stands mature and gaps occur in the canopy. Understory carbon represents only about 1% of total forest carbon, as described in Section 4.A. Though there

are only limited data to support the biomass trends applied here, related errors will have very little impact on stand-level analyses.

Forest floor carbon

Estimates of initial forest floor carbon and the age-specific increases in its pool size are based primarily on Vogt et al. (1986). Values rise gradually as the forest stand ages. The estimates for forest floor carbon of fully-stocked stands are also the same as used in Plantinga and Birdsey (In press). However, in FCM the pool size is adjusted proportional to the stocking level in the current inventory.

Soil carbon

To arrive at representative soil carbon pools for each forest type, geographic information systems technology was used to overlay the spatial distribution of each forest type with a map of soil carbon. Estimates of initial soil carbon within each region were based on the soil carbon map developed by Kern and Johnson (1991). In that study, data from over 3,000 pedons on record at the US Soil Conservation Service Lincoln Laboratory were used to estimate mean soil carbon at the great group taxonomic level. The geographic distribution of the great groups was derived from the 1982 National Resource Inventory (SCS 1987).

Forest-type distributions 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 and soils maps is relatively coarse. Nevertheless, the estimates of soil carbon pools for each forest type appear to be consistent with the geographic extent of species, with regard to climate, and earlier studies of soil carbon distribution in relation to climatic factors (Post et al. 1982).

The changes in soil carbon to be expected after stand initiation depend in part on stand origin. In the case of conversion from cropland or pasture to forest, there is likely to be a slow increase in soil carbon storage. The vast majority of current stands in the US originated from some form of forest disturbance, especially harvesting. 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. Recent studies in temperate forests do not indicate a significant decrease in soil organic carbon after tree harvest (Johnson 1992a, 1992b). Thus, in the FCM stand-level carbon budgets soil carbon remains constant. The net effect of our approach is comparable to Moore et al. (1981) in that the sum of carbon in soil, litter, and woody debris decreases significantly after stand initiation.

Examples of the stand-level carbon budgets which were developed using the above approach are given in Figure 3.1. 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.

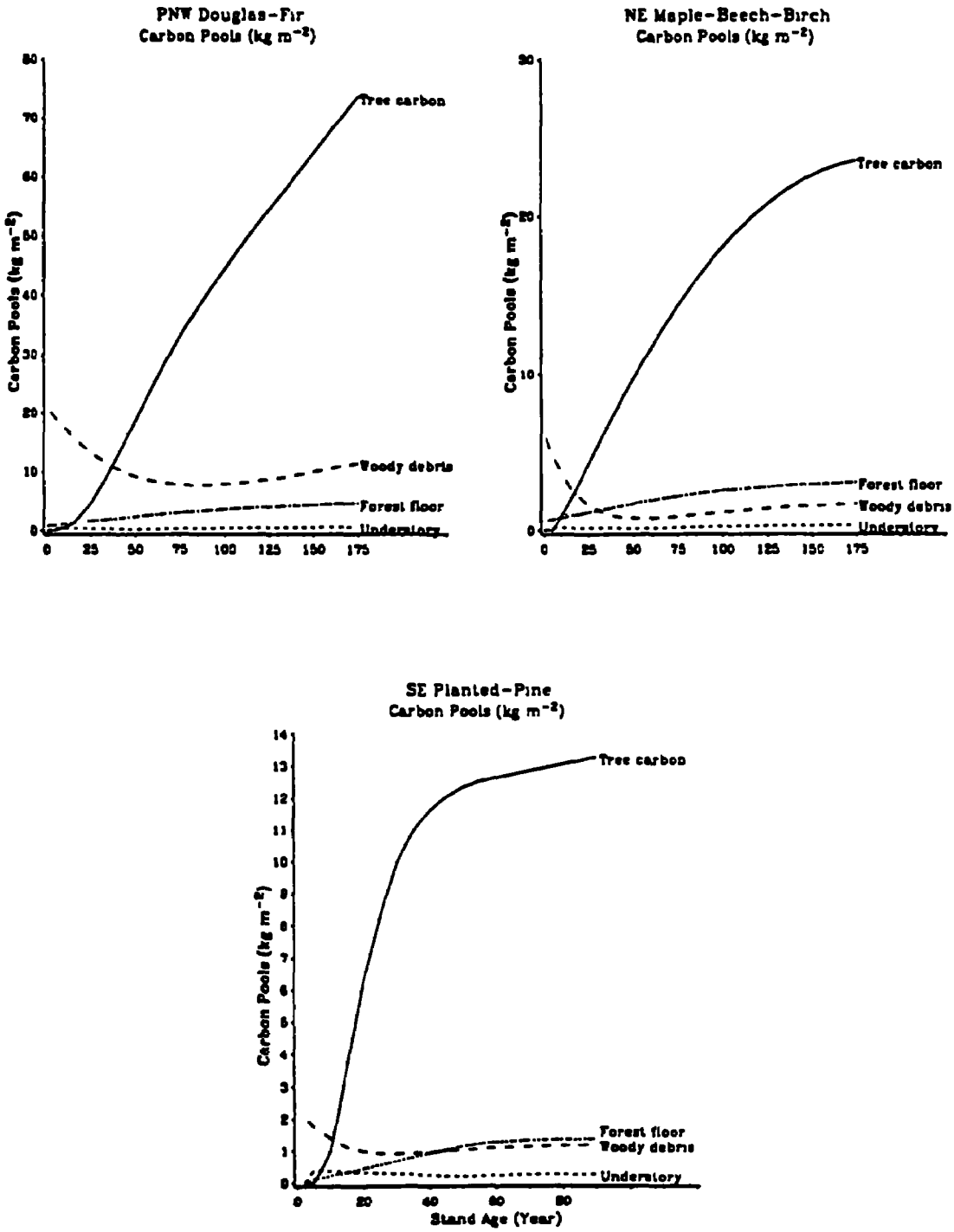

Figure 3.1. Examples of stand-level carbon budgets for three forest types using the FCM. Abbreviations are as follows: PNW = Pacific Northwest, NE = Northeast, and SE = Southeast.

2. Forest inventory databases

Private lands

A dataset for the forest inventory on private lands in 1990 (the base year) was generated by the coupled TAMM/ATLAS model described in Section 3.C. The survey data used to generate the forest inventory input to ATLAS was the same survey data used to compile the 1989 RPA Assessment (Mills 1988, Haynes 1990) and includes information on forest type, stand age, ownership, productivity level, and timber management intensity. There is a direct linkage of ATLAS inventory categories to specific stand-level carbon budgets based on ATLAS yield tables. ATLAS also carries information on current growing stock volumes.

For the eastern US, the RPA survey data was collected by the USDA Forest Service FIA work units at state and county levels for all ownerships. In contrast, data collected by the FIA work units at state and county levels in the western US have been limited to non-federal lands. A double sampling scheme (Cochran 1977) is used in the FIA surveys, whereby owner group, land class, and forest condition class are identified on photo points and the photo sample is subsampled with a grid of field plots. The field plots are the basis for information on timber volume, growth and mortality, and attributes such as forest type, site class, and stand-size class. Uncertainty in these surveys related to sampling intensity and frequency are discussed in Section 9.A.

Public lands

The inventory data developed for ATLAS contained age class information for private lands only. To estimate carbon pools on public timberland, it was necessary to generate age class specific area and volume data which could be processed with the stand-level carbon budgets previously described.

Information on the areal extent of forest types on public lands and the total growing stock volume was available from the 1987 RPA database. Data on the age class distributions on public lands within each forest type exist for some particular areas, but have not been compiled on a national scale. Additional inventory work and database augmentation would improve future analyses. Satellite remote sensing has begun to be applied for this purpose (e.g., Tepley and Green 1991); however, there is as yet no published systematic treatment of public lands.

Lacking a national inventory for age-class distributions on public lands, several assumptions were employed for the purposes of this initial carbon budget development. Within the Southeast, South Central, Northeast and North Central regions (Figure 3.2), the area of public timberland is relatively small (Figure 3.3). 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 volume of public lands reported in the 1989 RPA Assessment by forest type and region (Waddell et al. 1989) was allocated among age classes in direct proportion to the ATLAS inventory data for private lands. For each forest type represented on public lands, the corresponding stand-level carbon budget from ATLAS was applied regionally.

In the Rocky Mountain, Pacific Northwest East, and Pacific Southwest regions, the area of public timberland is quite large; however, the average

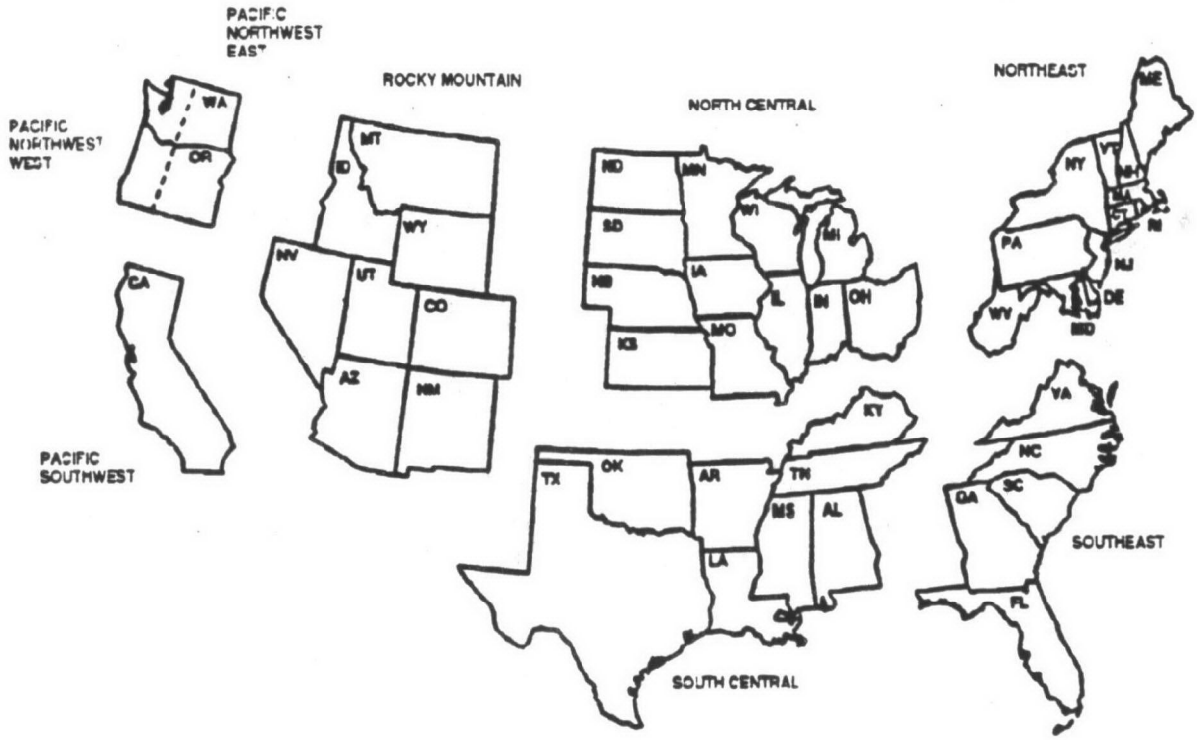

Figure 3.2. Definitions of regions used in assessment of the US forest sector carbon budget. Abbreviations are as follows: PNWW = Pacific Northwest West, PNWE = Pacific Northwest East, PSW = Pacific Southwest, RM = Rocky Mountain, SC = South Central, SE = South Southeast, NC = North Central, and NE = North Northeast.

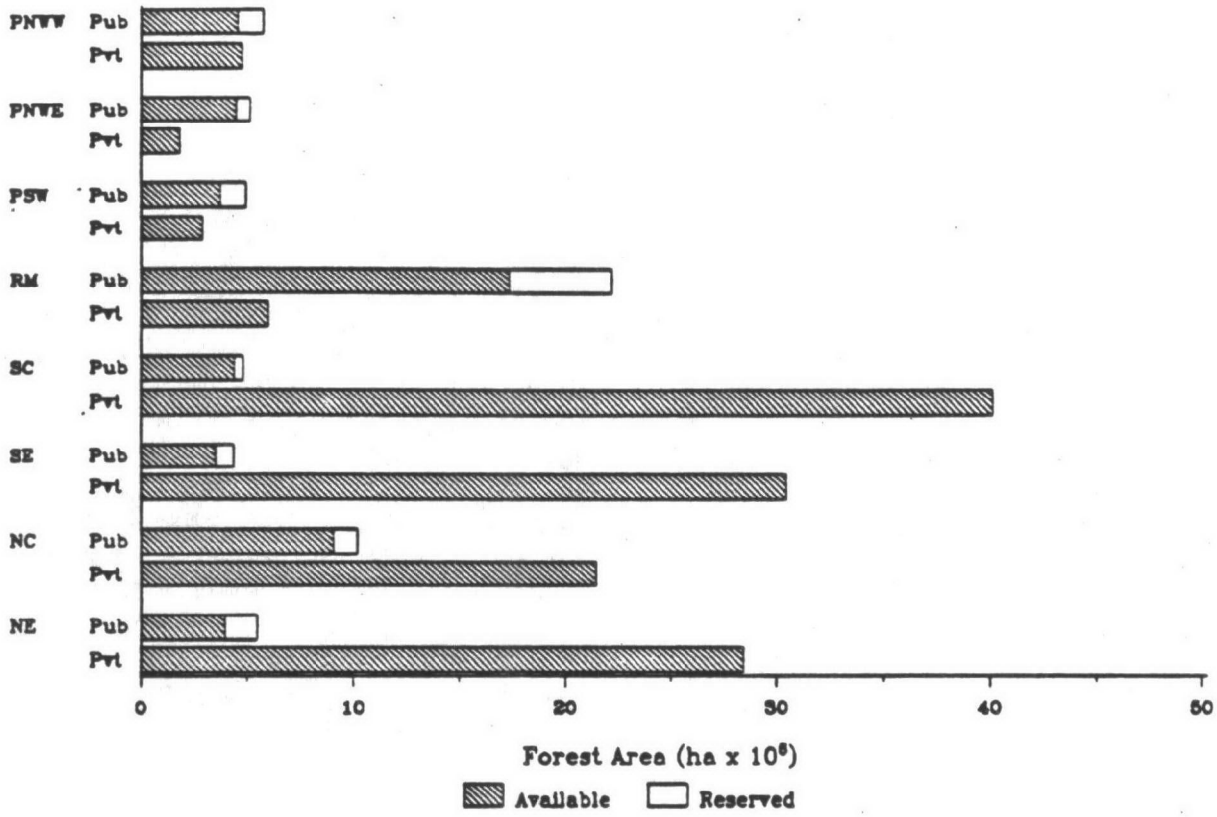

Figure 3.3. Base year distribution of available and reserved timberland by region and ownership. Abbreviations for regions are the same as in Figure 3.2, Pub = public ownership, Pvt = private ownership.

volume per unit area is low relative to other regions. Lacking any consistent or 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. Stand-level carbon budgets were similarly chosen.

In the Pacific Northwest West region, recent state level studies collated age class data for national forests and other public lands (Oregon, Sessions 1991; Washington, Adams et al. 1992, MacLean et al. 1991). This information was used to estimate age class distributions and stocking levels on public lands and, as before, the stand-level carbon budgets were used for estimation of carbon pools. 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 greater than 175 years were assigned to the 175-year age class.

B. Base year carbon flux

The reference state in this analysis must be the net accumulation or loss of carbon from the forest land base over a one year period due to factors associated with the forest sector. The term flux refers to carbon transfer between the forest and the atmosphere in either direction or 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. We have also estimated the change in carbon stored in forest products which are still in use and still in landfills.

1. Biologically-driven carbon flux

The biologically-driven carbon flux is an estimate of the change in total carbon storage expected over a 1-year period assuming there had been no harvest. This flux for individual forest components, and total flux which reflects net changes in all the pools, 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 simply the change in the carbon pools over a specified interval, divided by the number of years in the interval (e.g., 5 years in the Southeast region and 10 years in the Pacific Northwest). Initial stocking levels and expected adjustments to the stocking levels, which are prescribed by ATLAS, were allowed to impact uptake rates. Within each forest type, the area within each age class of each relevant stand type was then multiplied by the associated carbon flux rate per unit area. The estimate of annual carbon uptake over a region was an aggregate of the uptake for all forest types within that region.

The approach used here does not account for woody debris decomposition on lands which have been deforested and are no longer in the forest inventory. This factor may not be large in the US 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 CO_2.

2. Harvest-driven carbon flux

In addition to the biologically-driven carbon flux, the flux resulting from harvesting and commercial thinning was also considered. For the areas harvested in the base year, an accounting

was made for the carbon in all pools.

The volume of growing stock and non-growing stock removals on private and public lands for the base year was derived from the 1989 RPA Assessment and is comparable to the removals reported in Waddell et al. (1989) and Haynes (1990). Conversions from volume to carbon were made using the factors in Table 3.1. The harvested carbon was removed from the forest inventory and represented a large offset to the biologically driven flux. The residue from harvest which is left on site was modeled as a transfer of carbon from the living tree biomass to the woody debris pool. Thus, there was no effect on total carbon storage. In practice some of this carbon might be burned as slash; however, rather than attempt to account for rates of slash burning, we have let the residue decay as part of the woody debris pool. This approach provides assurance that this carbon is consistently accounted for.

The change in carbon in the understory and the forest floor of harvested stands was treated as instantaneous since it consists of rapidly decomposing material. Thus there was a small carbon pulse from the forest land base associated with the reduction of understory and forest floor in the harvested areas. In order to estimate this carbon pulse, the distribution of the harvest among forest types and age classes was needed. On private lands the harvest volume was distributed across forest types and age classes by ATLAS (described in Section 3.A). On public lands, the RPA Assessment (Haynes 1990) provided harvest volume by region but no information on the distribution among forest types and age classes.

For this analysis, we assumed an equivalence in the distribution of harvest across forest types and age classes between public and private lands, except for Pacific Northwest forests west of the Cascades. In that region, we used data from Sessions (1991) on the distribution of the harvest among age classes on public lands.

3. Other changes in carbon pools

If none of the harvested carbon returned to the atmosphere, it could be considered as a carbon sink in addition to whatever net change occurred on the forest land base. However, this carbon is returned to the atmosphere eventually, much of it rather rapidly. The actual sink associated with the forest harvests is only the change in the pool composed of products still in use and products in landfills but not yet decomposed or burned.

The change in the size of the two products pools is a function of annual inputs and outputs. The annual change in these pools was evaluated by reference to literature-based estimates of the proportion of the harvest going into products, the annual transfer of wood products into landfills, and the annual carbon flux out of landfills.

C. Projected US forest sector carbon pools and flux

The approach to estimating future carbon pools and flux relies on inventory projections provided by models which focused on forest economics and inventories. The inventory at any point in time is run through the Forest Carbon Model to estimate total forest carbon. The change in total forest carbon over a 10-year interval, divided by 10, gives the average annual flux for that interval.

1. Use of the TAMM and ATLAS models for private timberlands

Timber is a unique commodity in that its production period requires more long-term planning than virtually any other renewable resource. Informal assessments of the timber situation date from 1866 in the US. These were formalized by the Forest and Rangeland Renewable Resources Planning Act of 1974 (RPA) which calls for periodic long-range assessments of timber supply and demand to provide information for stewardship and management decisions in both the public and private sectors. Starting in the 1960's, the USDA Forest Service began developing more formal planning frameworks for timber assessments.

The basic approach was a trend analysis of both demands for forest products and availability of timber resources (USDA Forest Service 1965, 1974, and 1982). Policies and emerging trends were discussed in the context of the gap between the trajectories of the demands for forest products (expressed in roundwood equivalents) and the prospective availability of timber. In 1977, the USDA Forest Service undertook the development of an assessment model designed to explicitly explain regional stumpage price behavior. This model was built around a spatial equilibrium algorithm.[1] The initial purpose was to develop regional harvest and price trends for the 1979 RPA Timber Assessment. The resulting model, called TAMM, exceeded expectations (Adams and Haynes 1980, Haynes and Adams 1985). It produced regional price trends in both product and factor (stumpage) markets that recognized simultaneous market interactions and differences in regional timber resources. This model built upon early econometric studies and linear programming efforts that attempted to solve spatial market concerns.

TAMM provides an integrated structure for considering the behavior of regional prices, consumption, and production in both stumpage and product markets. It is designed to provide long-term projections of price, consumption, and production trends and to simulate the effects of alternative forest policies and programs. To a far greater extent than was possible in the past, the use of TAMM has focused attention on the dependence of projections on (exogenous) input assumptions. In Forest Service assessments these assumptions concern the major determinants of the supply and the demand for various forest products.

This section provides broad details about how TAMM operates. Details about the various input assumptions used in TAMM are described briefly in the next section and in more detail in the 1989 RPA Timber Assessment (Haynes 1990).

The general structure of TAMM is shown in Figure 3.4. It is composed of ten major parts with exogenous inputs from four other models (Pulp, Trade, Fuelwood, and Area Change). Imports are explicitly treated by including Canada in the model. Briefly, product demand, such as softwood lumber, is obtained by multiplying the ratio of

[1]The emphasis on regional markets meant that the model had to deal with the spatial juxtaposition of stumpage markets. Samuelson (1952) is generally credited with defining the problem, but until the last decade its solution has always involved complex computational methods. Previous experience with linear programming models, both in agriculture and forestry, led model builders in the US to select a computationally simpler algorithm (a Reactive Programming algorithm developed by King and Ho [1972]) to solve for spatial equilibrium.

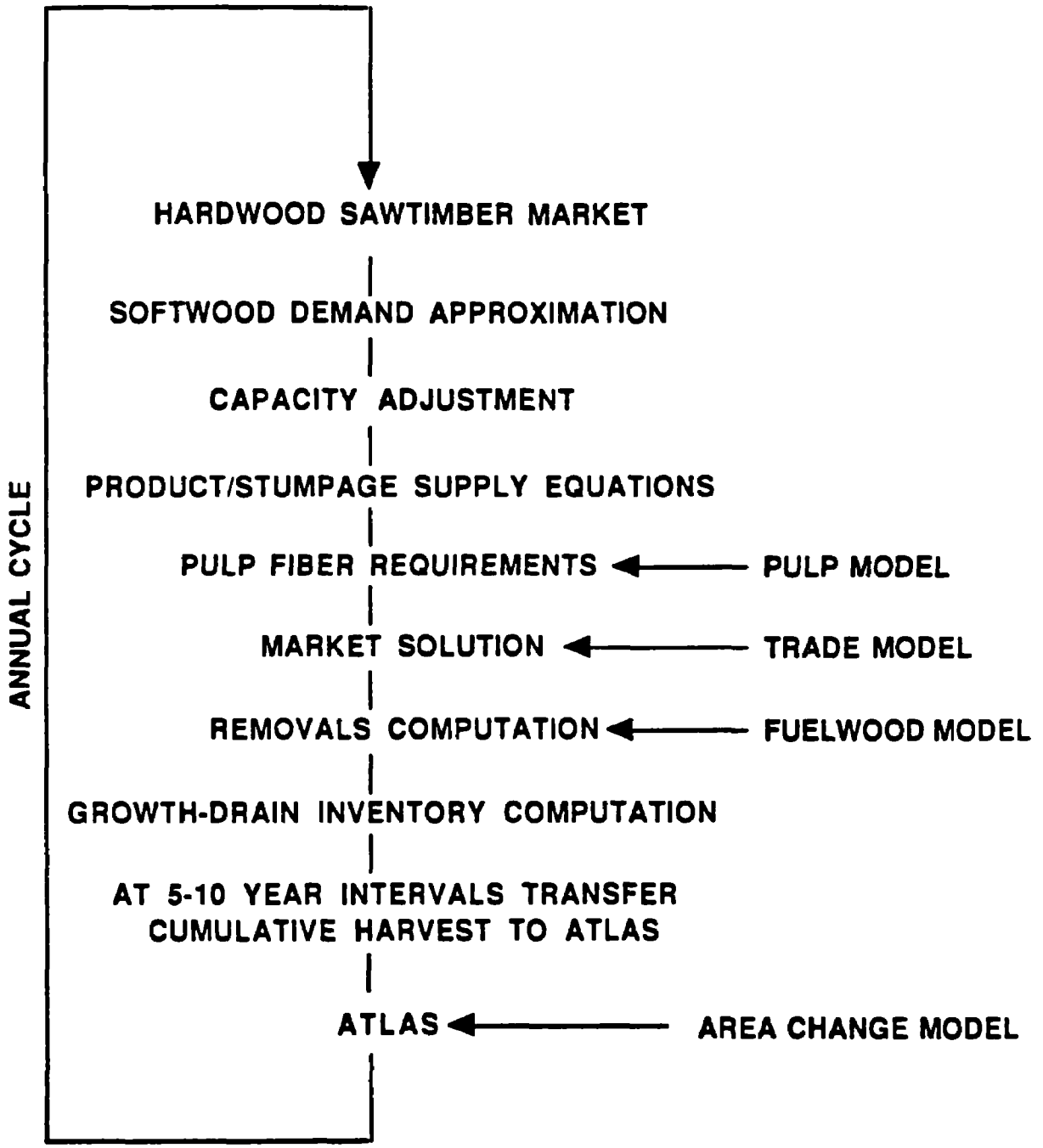

Figure 3.4. General structure of TAMM.

product use per unit of activity (such as the number of housing starts) times the number of units and summing these results over all the various end-uses for the product. The activity measures are exogenous and are generally taken from long-term macro forecasts prepared by Wharton Econometrics Forecasting Associates (1987). An example demand function (ignoring the time subscript) would be given by the expression:

$$C_i = \sum_u F_{i,u}(P_i, P^{s_{i,u}}, Z_u) * I_u \qquad (1)$$

where:

C_i is the consumption for product i,

$F_{i,u}$ represents a system of equations whose solutions give the end use (u) factors as a function of their several arguments,

P_i the price of product i,

$P^{s_{i,u}}$ is a vector of prices of substitute materials,

Z_u is a vector of other shifters, and

I_u is a measure of activity in end use u.

The demand relationships are regionalized by accounting for differences in regional per capita consumption and the ratio of regional prices to national prices. Hardwood lumber is treated on the demand side in about the same end use detail as softwood lumber but consumption and production are set equal and price is determined as a function of softwood lumber prices. Hardwood sawtimber[2] stumpage prices are a function of hardwood lumber prices.

The product supply equations are estimated in the form:

$$s = (p - avc)\, k \qquad (2)$$

where

s is the supply volume,
p is the product price,
avc is the average variable cost, and
k is the installed capacity.

Average variable costs are composed of wood and nonwood elements that have been converted to a unit output basis by means of product recovery and productivity factors that reflect the current (or projected) technology of the industry. Each product supply function includes installed capacity as an independent variable. Shifts in installed capacity are modeled as a function of anticipated changes in relative regional profitability or rate of return. Timber supply is modeled as a function of stumpage price and timber inventory. Finally, the timber demand functions are derived from product market demand and supply functions. The pulp fiber requirements are input variables from a pulp model developed by Ince, Durbak, and Howard (in press). Trade and fuelwood projections are also input variables[3]. The trade projections reflect a future where the US remains a net importer of softwood forest resources.

The remaining steps in the annual cycle

[2] Sawtimber stumpage includes live trees of commercial species containing at least one 12-foot sawlog or two noncontiguous 8-foot logs, and meeting regional specifications for freedom from defect. Softwood trees must be at least 9.0-inches dbh, and hardwood trees must be at least 11.0-inches dbh.

[3] See the 1989 RPA Timber Assessment for further details (Haynes 1990).

of TAMM involve timber supply and an inventory projection system (ATLAS).[4] The methods used to project timber supplies require a number of assumptions that include the form of the inventory projection system, projections of timberland area change, assumptions linking harvest to removals from inventory, and projected harvest flows from public timberlands. The basic economic representation of timber supply describes the supply of timber at any point in time as being a function of the private timber inventory levels, stumpage prices, and the amount of public harvest available at that time.

The Aggregate Timberland Assessment System (ATLAS) was used to make the inventory projections on all private timberland in the US. It is an age-based, yield-table system that projects acres by detailed strata for periods consistent with the inventory stand-age classes. It evolved from earlier work by Beuter et al. (1976) and Tedder et al. (1987). In the model, the inventory is represented by acreage cells classified by region, ownership, management type, management intensity, and age class. In the South and the Pacific Northwest West, the strata were also identified by site productivity class and stocking class.

[4]This inventory projection system considers only the timber inventory located on timberland that is producing or is capable of producing crops of industrial wood and not withdrawn from timber utilization by statute or administrative regulation Currently, areas qualifying as timberland have the capability of producing in excess of 20 cubic feet per acre per year of industrial wood in natural stands and may be inaccessible and/or inoperable. Further, these models include only live trees of commercial species meeting specified standards of quality or vigor. Cull trees are excluded. When associated with volume, these growing stock inventories include only trees 5.0-inches dbh and larger. The ATLAS model was developed by Mills and Kincaid (1992) for the 1989 RPA Timber Assessment.

This study used 10-year age classes in all regions except the South where 5-year age classes were used.

The ATLAS model starts with data arrayed in the following fashion for each relevant combination of region, forest type, ownership, site productivity, level and management intensity:

Age Class	Acres	Volume/ Acre	Yield Table
1	ac_1	CFA_1	YT_1
•	•	•	•
•	•	•	•
n	ac_n	CFA_n	YT_n

where:

ac_k is the inventory acreage of age class k

CFA_k is the average cubic feet of growing stock per acre (volume) inventoried for age class k

YT_k is the cubic feet growing stock per acre of a fully-stocked stand of age class k.

The basic time interval in ATLAS is set by how age classes are specified. If, for example, 10-year age classes are specified then the time dimension in ATLAS is in 10-year periods. That is, t is year 1 but t+1 is year 10. In the equations that follow, period 1 would be the first 10-years (except in the South where it would be the first 5 years).

ATLAS uses the basic growth-drain identity:

$$Inv_{t+1} = Inv_t + gwth_t - hvt_t \qquad (3)$$

where:

Inv_{t+1} is the inventory volume in period t+1,

Inv_t is the inventory volume in period t

$gwth_t$ is the growth in period t, and

hvt_t is the harvest in period t (This is an input variable to ATLAS)

Timber inventory as used here is the growing stock volume, defined as the cubic volume of all live trees of commercial species, meeting specified standards of quality or vigor, which are at least 5.0 inches in diameter (breast height).

The various variables are computed in a given period (t) as follows:

$$Inv_k = ac_k \cdot CFA_k \quad (4)$$
$$SR_k = CFA_k / YT_k \quad (5)$$
$$SR_{k+1} = a + b \cdot SR_k \quad (6)$$
$$ACHVT_k = (har_k \cdot hvt_t)/(CFA_k \cdot GH_k) \quad (7)$$
$$gwth_k = (YT_k \cdot SR_k) - (YT_{k-1} \cdot SR_{k-1}) \quad (8)$$

where:

SR_k is the stocking ratio for the k age class,

a,b are coefficients for stocking adjustments,

ACHVT are the number of acres harvested,

har_k is the harvest proportion for the k age class, and

GH_k is the growth-on-harvest proportion for the k age class.

Equation 6 adjusts the stocking percentages for stands over time, recognizing that stand density changes with time. This adjustment is called "approach-to-normality" and has been used in conjunction with growth predictions from normal (fully stocked) yield tables since the 1940's (Husch 1963).

In each simulation period, inventory change is the result of growth, area change, and timber harvest. Growth is the result of acreage following upward sloping net yield functions that are calculated with approach-to-normal assumptions. Each cell in the starting inventory may have an independent yield function. A major attribute of the model is that it can simulate shifts in management intensities, and the consequent changes in yields based upon alternative assumptions about the future. The inventory model is not, in principle, an even-age model since it can simulate growth and removal processes across all age classes. ATLAS can also account for both partial harvests and commercial thinning. Volume can be removed at harvest from either the oldest age class or spread across several age classes. Final-harvested acres can be assumed to be regenerated in any management type or may leave the timberland base entirely. Partial cutting affects a proportion of acres in a cell and the treated acres are assumed to return in the cell growth function to a younger age class.

Inputs to the inventory model include estimates of harvest, acreage shifts, and growth parameters. The levels of harvest are derived through interaction with TAMM. Area change information is supplied by a modified version of the Southern Area Model (Alig 1985). Yield tables and approach-to-normal parameters were derived from timberland inventory plot data collected by the various USDA Forest Service FIA units and from previous studies (the inventory data inputs are summarized by Mills [1988]).

The solution of TAMM represents a spatial equilibrium in the markets modeled for each year of the projection period. These solutions do not represent, nor can the basic market solution algorithm[5] be readily used to find, intertemporal production or consumption strategies that are in some sense optimal. The production, consumption, and price-time paths are only estimates of outcomes of contemporaneous interactions in freely competitive markets. Details about the various input assumptions used in TAMM are described in the 1989 RPA Timber Assessment (Haynes 1990).

2. Methods used in the analysis and treatment of public lands

As with information for the base year analysis, data representing projected changes in the public timberland inventory by age class are limited. For National Forests, process records from some individual National Forest plans contain projections by age class. Process records are part of the background information maintained by National Forests, as documentation required by planning regulations, and are used to monitor plan implementation. Many National Forest plans do not display complete sets of projections of harvest or timber inventory by age class, and in some cases the information is not readily extractable from models used in developing plans. In particular, different versions of the Forest Planning Model (FORPLAN) have been used across National Forests, which complicates the type of resource-related information provided in planning efforts. An additional complication is that National Forest planning and associated adjustments are an ongoing process. With significant developments such as the recent judicial rulings on spotted owl set-asides, harvest projections (and linked attributes such as the inventory level on timberland) for the National Forests are in a state of flux.

In the present study, the most recent projections of future removal rates and forest inventories on National Forests, as assembled by USDA Forest Service staff at the Pacific Northwest Station, were used for the scenarios described below. These were produced on a growing stock basis (m^3). The projections of timber inventories and harvest levels for Other Public lands (i.e. non National Forests) are the same for all scenarios and are taken from the RPA Assessment (Haynes 1990). Since only information on growing stock volume was available for the forest management scenarios, volume was used to estimate total carbon storage.

Unlike private lands, the conversion from growing stock volume to total carbon inventory on public lands was made on a regional basis by reference to the ratios of growing stock volume to total non-soil carbon (i.e., tree biomass, understory, forest floor, and woody debris). These ratios were taken from the base year analysis (Section 4.A). Soil carbon values were treated as constant in the scenarios because of the large uncertainties concerning variability of soil carbon in forests over time.

The projected timber volumes and total carbon on reserved lands were most problematical because inventory models do not exist for these lands. For this

[5] A revised reactive programming algorithm is used (Brooks and Kincaid 1987). Briefly, reactive programming is a method for solving continuous demand and supply function by successive adjustment of quantities produced and their distribution to demand regions, so as to maximize producer profits net of transport costs in each supply region.

analysis, the inventory on the unavailable public area for the base year was allowed to grow in 10-year increments and new carbon pools were calculated by running the inventory through the Forest Carbon Model.

The assumption of a fixed ratio of volume to total carbon has several limitations with regard to assessing the effects of harvest reductions. First, it does not account for age class distribution effects. The ratios of volume to woody debris and volume to forest floor carbon vary with stand age. A second problem concerns the treatment of changes in carbon storage in old growth stands. Because the yield tables which provide the foundation of the inventory approach to estimating carbon pools and flux reach a limit at 175 years, no increment occurs in these older stands. Thus, no change in carbon storage is reflected over a decadal period from all lands which remain in the oldest class. As discussed in Section 9.C, it appears that carbon does in fact accumulate in both living biomass and woody debris in these stands.

Because of the limitations in the preliminary USDA Forest Service inventory projections and in using ratios to estimate carbon from volume, an additional approach was taken to evaluate the effects of the Reduced National Forest Harvest Scenario described below (Section 3.C.4, Scenario 3). In that scenario, a greater than 50% reduction in the harvest was implemented for National Forests in the Pacific Northwest West region. The approach for assessing the consequences of this change was to determine the volume not harvested, to estimate from that volume the area not harvested, and to compare changes in carbon flux over time for this area on the basis of harvest and no-harvest scenarios.

Three potential differences in carbon flux were considered in this analysis. The first flux was the loss of the carbon sink which would have been maintained if the trees had not been harvested. The magnitude of this sink tends to drop after the rotation age. But even 50 years after the rotation age, it is still over half of the maximum rate achieved early in the rotation for Pacific Northwest West Douglas-fir stands. The second flux was the on-site carbon source created by the harvest. As discussed in Section 3.B, harvested stands are carbon sources for about the first 20 years in the Pacific Northwest West because the decay of woody debris dominates the carbon uptake associated with tree growth. During the next 30 years these harvested areas begin to accumulate carbon. Lastly, the off-site carbon source which would have been created by the decay of relatively short lived wood products derived from the harvest was estimated. The carbon which would have accumulated in forest products in and out of landfills is assumed to be balanced by carbon emissions from landfills.

Changes in volumes harvested were taken from the TAMM/ATLAS run, with the 1989 RPA Assessment as the reference harvest level. The initial year of the analysis was 1995, when the major harvest reductions were in place. The land area harvested per unit volume harvested was based on a ratio derived for National Forest lands in Washington and Oregon. Data were available from several sources for harvest by forest type and age class in these states (Sessions 1991, MacLean et al. 1991). The area not harvested was then assigned to a stand-level carbon budget associated with Douglas-fir on a medium productivity site, as described

in Sections 3.A and 3.B.

The carbon sink associated with the unharvested area was modeled through the year 2045. During each year of the simulation, the new unharvested acres were added to an inventory, with an age class corresponding to the rotation age (80 years). Then the time course for the total carbon flux from that time forward was used to estimate the carbon sink associated with that area, assuming it had not been harvested. Thus, over the 50-year simulation, the carbon increments for the age classes from 80 years to 130 years were used.

To estimate the on-site carbon source that was avoided by not harvesting, an area equivalent to the unharvested area was added to a second inventory with an age class corresponding to the youngest age class, thus simulating harvest. The first 50 years of the same Douglas-fir stand-level carbon budget was used to estimate the biologically-based carbon flux which would have been generated after the harvest.

The off-site carbon source associated with decay of short-lived harvested wood products, which would be avoided, was estimated at 58% of the volume harvested. This is an approximation of the proportion of the harvest in the Pacific Northwest which returns relatively quickly to the atmosphere (Harmon et al. 1990). There may be compensatory increases in harvest that might occur on private lands but that has not been considered here.
The net effect of not harvesting was calculated as the sum of the on-site sink which was maintained, the on-site source which was avoided, and the off-site forest product source which was avoided. Each year over the 50-year simulation, the net effect for all areas not harvested relative to the RPA Scenario was summed for the total net effect.

3. Fate and disposition of carbon after timber harvest

During processing, use, and eventual disposal, all the carbon harvested from forests is eventually returned to the atmosphere as carbon dioxide or methane by either burning or decay. However, a large proportion of harvested carbon may remain in use in products for varying lengths of time, or accumulate in landfills where decomposition rates are slow. To the degree that the products and landfill pools are increasing or decreasing, a sink or source of carbon is created. It is important to account for these carbon cycle flows because their impact on the overall US forest-sector carbon budget could be significant.

The model HARVCARB (Row and Phelps 1990) was used to examine distribution and transfer of post-harvest carbon quantities predicted by the TAMM/ATLAS analyses of policy alternatives. The model traces the flow of forest harvest carbon through five phases of processing, use, and disposal:

- From trees to roundwood logs taken from the forest.
- Through processing into forest products like lumber and paper.
- Into uses in construction, manufacturing, and paper products.
- To discard products (construction debris and municipal solid waste).
- Finally, to landfills, incinerators, and ultimately, back to the atmosphere.

The first three phases occur within a

year or two. Once the products are manufactured, some might remain intact for years or decades, and might be replaced or abandoned at a slow rate, such as construction timber. Others, such as most paper products, may be used and discarded promptly. After being discarded, the materials might reside in landfills and debris dumps for long periods.

The results of the HARVCARB analysis pertain only to carbon harvested during the analysis period beginning in 1980 and not to any carbon that was already in products and landfills. It is difficult to assess the actual incremental changes in these pools because not only are their pre-1980 sizes unknown, but the fluxes of pre-1980 carbon between products, landfills, and atmosphere are also unknown. Some indication of the impact of any given scenario can be obtained by comparing the differences between it and the reference scenario in the carbon stored in products and landfills.

4. The forest-sector scenarios

In this report we consider 12 forest sector scenarios (Table 1.1). The scenarios ranged from the Cureent Forest Plans, which is the base scenario and reflects current best estimates of future harvest levels on public lands, to scenarios that called for reduced harvests on public lands, increased recycling, or increased afforestation. The 1989 Resources Planning Act (RPA) scenario was included because it represents the most recent well-documented scenario produced by the USDA Forest Service. Differences between the RPA and the Current Forest Plans scenarios are a function of policies already in place. The last three scenarios consist of combinations of the assumptions associated with some of the other nine scenarios. Each scenario was run using the combined TAMM/ATLAS/FCM modeling framework (Figure 1.2).

Scenario 1. The 1989 RPA Assessment Projection

The assumptions and projections of the 1989 RPA Assessment are discussed in detail by Haynes (1990). For private lands these assumptions are basically the same as in the Current Forest Plans Scenario, described subsequently.

The background assumptions for any TAMM model run include projections of future forest products markets (based on forecasts of the major determinants of product demand, and of timber and product supply). On the demand side, the major determinants are future growth in aggregate economic activity, as measured by gross national product (GNP), and on certain key end-uses for forest products, such as new residential construction/repair and alteration of existing dwellings. In the RPA Scenario, the population of the United States is expected to grow from 242 million in 1986 to 333 million by 2040, and considerable improvement is expected in worker productivity. The result is projected GNP growth rates of 2 to 3% over the next 50 years. This portrays a strong and resilient future economy, with an increasingly affluent population. It suggests continued strength in demand for forest products, such as paper and paperboard, that are used in a wide range of sectors in the economy.

Residential construction particularly consumes solid wood products. Projections of the factors that determine long-term demands for new housing units (household formations, replacement of units lost from the housing stock, and the inventory of vacant units) indicate continued high levels in the early 1990's with about 2.0 million

starts of all types of housing per year, decreasing gradually after 2000 to about 1.5 million starts per year. About 20% of the lumber and structural panel products and 15% of nonstructural panel products are used each year for the upkeep and improvement of existing units. Residential upkeep and repair have grown rapidly during the 1980's and are expected to continue growing as the nation's housing inventory and the average age of its component units increase. Key supply determinants include levels of public harvest, changes in manufacturing and logging technology, and trends in private timberlands.

Scenario 2. Current Forest Plans

This scenario reflects current (1992) knowledge about future timber supply, demand and harvest on private and public lands. As discussed below, it includes some reductions in harvest on public lands due to habitat preservation and transfer of lands from available to reserve classification. It is currently considered to be the "base scenario" for USDA Forest Service planning. Because it differs from the RPA Assessment Scenario, it is described in detail below.

In the last two decades, it has become increasingly clear that the future of timber production on the National Forests depends in part on (1) success in finding suitable ways to integrate timber production with other uses of forest land, and (2) the need to protect and maintain the forest environment, including endangered and threatened species. The controversy surrounding habitat protection for old growth species such as the spotted owl illustrates the increasing constraints on timber production on the National Forests. The Current Forest Plans and the Reduced National Forest Harvest scenarios are presented to address these two issues. The total National Forest harvest levels for each case are shown in Table 3.2.

The 1989 RPA Assessment harvest projection essentially represents a continuation of the late 1980's harvest levels. The increases after 2000 come from modest increases of planned harvests in the Rocky Mountains. The National Forest harvest projections for the Current Forest Plans scenario projection assume the adoption of the Interagency Scientific Strategy for Protection of the Northern Spotted Owl (ISC 1990) and the most recent National Forest Plans (on file with each National Forest). Additional factors which are considered include a more prolonged recession and changes in log export restrictions for Western states.

For the Current Forest Plans scenario, National Forest softwood harvest is projected to fall from the recent levels of 2.2 billion cubic feet to about 1.7 billion cubic feet by 2000 and then rise to 2.0 billion by 2040. All of the downward adjustment in the three Pacific Coast states results from the adoption of both final forest plans and the conservation strategy for the northern spotted owl (ISC 1990). The combination of these two changes, for example, is to reduce National Forest harvest in the Douglas-fir region by 51%. National Forest hardwood harvest remains relatively stable at about a billion board feet per year. Harvests from other public timberlands are projected to increase as those inventories increase.

Projections of technological trends for the Current Forest Plans scenario are described by Skog et al. (1991) and changes in the private timberland base are described by Alig and Wear (1992). In spite of the numerous changes, the

Table 3.2. Projected harvests (as growing stock) from National Forests for RPA, Current Forest Plans, and Reduced National Forest Harvest scenarios.

Year	RPA Assessment Projection	Current Forest Harvest Plans	Reduced National Forest Harvest
	billion cubic feet		
1986	2.16	2.16	2.16
2000	2.17	1.72	1.36
2010	2.32	1.82	1.42
2020	2.40	1.89	1.47
2030	2.47	1.96	1.54
2040	2.53	2.01	1.59

general outlook remains roughly the same suggesting that the 1989 RPA Assessment projections are robust both as a base for policy actions and as a view of the future.

Assumptions about markets and prices for manufactured products were as follows:

Lumber. Total lumber consumption in 1986 was 8.84 billion cubic feet.[6] Consumption of lumber is projected to rise, reaching 10.58 billion cubic feet in 2040, as the use of softwoods in construction (particularly upkeep and alteration) and hardwoods in manufacturing and shipping continue to increase (Figure 3.5). After 2020, declining consumption in housing is more than offset by continued growth in manufacturing and nonresidential uses for both hardwoods and softwoods. National Forest harvest reductions lead to increases in the level of Canadian lumber imports, but in the longer term rising delivered wood costs in Canada lead to a decline in imports to 1.98 billion cubic feet by 2040. Hardwood consumption also increases steadily and the sector continues to be a net exporter.

Lumber production shares by region for 1952, 1992, and projected years are listed in Table 3.3.

Softwood lumber production has shifted among US regions during the last four decades and is expected to continue to shift. In 1992, the South and the Pacific Coast states accounted for 80% of the softwood lumber production (down from 86% in 1952). During the 1950's and 1960's, the three Pacific Coast states were the major lumber producing region, but starting in the 1970's the lead transferred to the South. Softwood lumber production is projected to shift from the Pacific Coast regions to the South over the next five decades, due to continued growth in stumpage costs

[6] All volumes reported for manufactured products in this section are in roundwood equivalent.

Table 3.3. Lumber shares (%) by region for 1952, 1992, and projected.

	1952	1992	2000	2040
	----------Percent----------			
Pacific Coast	57	46	44	37
Rockies	8	14	15	15
North	6	6	6	8
South	29	34	35	41

in the Douglas-fir and Pacific Southwest regions. Softwood lumber production in the northern regions and in the Rocky Mountains rises throughout the projection period. The expansion in production in the Rocky Mountains results from increased regional National Forest harvest levels which lead to decreased stumpage costs relative to other regions.

Structural Panel Products. Structural panels (softwood plywood and oriented strand board, OSB) consumption rises to 2.56 billion cubic feet in 2040, about 50% above 1986 consumption. Most of the increase is due to continued growth in OSB, which is projected to reach 1.0 billion cubic feet by 2040, more than 3.6 times its use in 1986. After slowly declining through 2010, softwood plywood consumption increases to 1.6 billion cubic feet in 2040. By 2040, OSB will compose over 40% of total structural panel consumption, up sharply from 16.6% in 1986. Consumption of panels is expected to increase across all end-uses except for new housing.

Regional production of structural panel products has undergone major changes in the past. Southern softwood plywood output has grown to 51% of total production and the North has emerged as a major producer of OSB structural panels. Continued regional shifts are projected in the next 20 years. By 2000, softwood plywood production will decline both in the South Central and Douglas-fir subregions. OSB production will nearly double in the North during the same period. Between 2000 and 2040, panel production for both softwood plywood and OSB expands in all regions except for the Rocky Mountains and regional production shares stabilize.

Nonstructural Panel Products. Consumption of nonstructural panels, including hardwood plywood, insulating board, hardboard, and particleboard was 0.77 billion cubic feet in 1986. It is projected to rise to 1.13 billion cubic feet by 2040. Different trends in demand are projected for the various products. Little growth in insulating board, whose major market is residential construction, is expected. Hardwood plywood, used in manufacturing as well

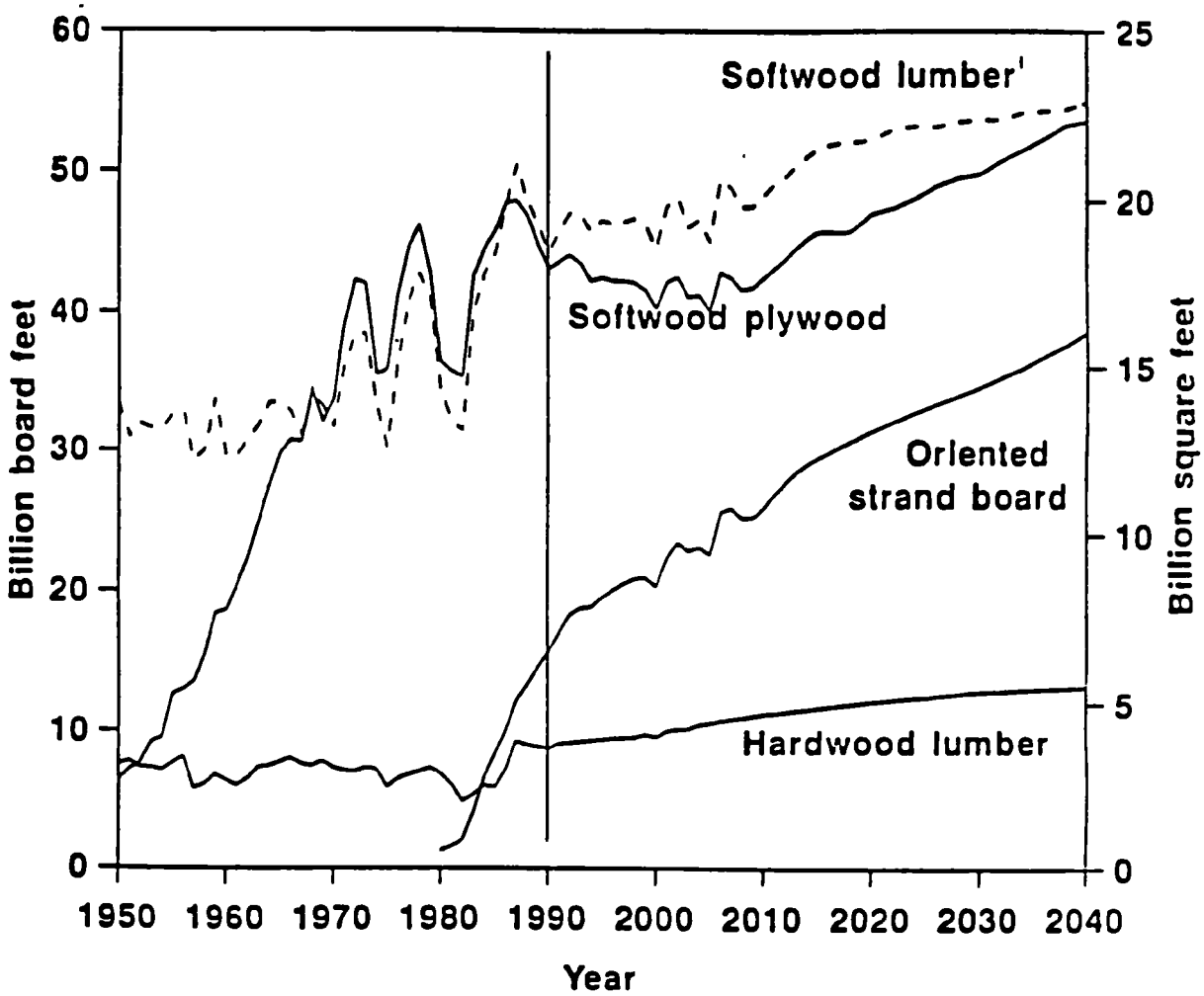

Figure 3.5. Lumber and structural panel products consumption, 1950-1988 with projections to 2040.

as construction, increases slowly through 2040, while particleboard rises until 2010, but shows little growth afterwards. Hardboard is the only nonstructural panel product to show a steady increase throughout the projection period.

Fiber Products. Consumption and production of paper and board is projected to increase, although not as strongly as in the past. Total consumption will exceed 100 million tons in 2000 and 173 million tons by 2040. Imports exceed exports over the projection period, with production approaching 166 million tons by 2040. Growth in per capita consumption of paper and board slows, reflecting substitution of non-paper for paper materials in packaging and construction products and less rapid expansion in the consumption of white paper.

Woodpulp (the major input for paper and board manufacture) is projected to increase 93% between 1986 and 2040 to 116 million tons. Imports of woodpulp are projected to continue to exceed exports. At the same time, consumption of wastepaper is projected to triple, reaching 46 million tons per year (about 29% of total fiber consumption). The projected expansion in wastepaper use is sizable by historical standards in the US, but the 2040 level is still quite modest compared to other industrialized countries.

Pulpwood consumption grows less rapidly than use of woodpulp due to higher yields from pulpwood and changing species mix. Hardwood pulpwood consumption increases from 2.2 billion cubic feet in 1986 to 5.1 billion cubic feet by 2040, an increase of more than 130%. Softwood pulpwood consumption is projected to increase by 50% from 1986 to 2040. The higher rate of increase for hardwood pulpwood reflects a gradual shift in the industry toward high-yield mechanical pulps.

Miscellaneous Timber Products. A variety of other industrial timber products, including poles, piling, posts, log exports, round mine timbers, bolts used for shingles, handles, and woodturnings, and chemical wood, are produced or consumed in the US. Consumption of roundwood in these products will rise slowly from 0.8 billion cubic feet in 1986 to 0.9 billion cubic feet by 2040. The softwood log export component of this category (mostly derived from Washington, Oregon, and California) is projected to decline to roughly 0.5 billion cubic feet by 2000 and remain at that level through 2040.

Fuelwood. Total fuelwood consumption, which in 1986 was an estimated 3.1 billion cubic feet, is projected to increase to 5.4 billion cubic feet in 2040 with most of the increase occurring before 2010. This is a result of projected cost advantages during this period for fuelwood relative to nonwood fuels for industrial, commercial and residential uses. After 2020, with moderating growth in nonwood fuel costs, fuelwood use in residences declines.

Product Price Projections[7]. In the Current Forest Plans projection, softwood lumber prices continue to grow for the next two decades at about the long-term average (1.4% per year), reflecting declining sawtimber[8] in-

[7]All prices are measured in 1982 dollars. This excludes the effects of general price inflation or deflation. The increases shown, therefore, measure change relative to the general prices of most competing materials.

[8]Here we are using a more general definition of sawtimber that includes the part of harvest being used in the manufacture of lumber, plywood, and miscellaneous products, and as log exports

ventories on private timberlands in both the West and South (Figure 3.6). After 2010, however, major increases in softwood sawtimber inventories roughly stabilize product prices (see Table 3.4). Over the next five decades, average softwood price growth is 0.7% per year. Hardwood lumber prices are expected to rise at an average annual rate of about 0.9% per year, reflecting continued growth in major end-uses and a steady decline in the availability of larger timber for higher quality lumber grades. Projections (Table 3.4) also show rising real prices for plywood in the near-term but roughly constant prices for OSB. Efficiency improvements lead to little change in pulp and paper prices. We expect that future price trends will mirror the experience of the period 1962-1976, when cost reductions due to technical changes allowed stable or declining real prices.

Projections of Timber Markets and Inventories. Given the projections of total product demand and net trade, harvest from US timberlands will increase sharply over the next five decades, rising some 50%, from 18.0 billion cubic feet in 1986 to 27.1 billion cubic feet in 2040. Hardwood harvest is expected to rise more rapidly than softwoods. Between 1986 and 2040, hardwood cut is projected to rise about 79% to 11.3 billion cubic feet, while softwood cut grows by some 35% to 15.8 billion cubic feet.

Projections of sawtimber softwood stumpage prices are shown in Figure 3.6 for selected regions. Softwood prices rise substantially in all regions. Projected growth rates over the 1986-2040 period are: 2.0% in the Pacific Coast region, 3.2% in the Rockies, 1.5% in the South, and 2.5% in the North. In all regions, price increases are most rapid in the period up to 2010 with a clear slowing in growth (or even decline in some regions) in the last three decades of the projection. Growth in eastern hardwood prices averages 1.4% per year.

The time patterns of timber prices shown in Figure 3.6 reflect both the underlying assumptions regarding public harvests and projected changes in the volumes and ages of timber inventories on private lands. For softwoods, the rapid price growth to 2010 results from limitations on timber supply in both the Pacific Coast and Southern regions over the next two decades. Between 1990 and 2000, though public cut is assumed relatively stable, industrial private harvest in the Pacific Coast region falls in the face of dwindling volumes in merchantable ages and sizes. This forces timber, and ultimately solidwood product, prices upward and leads to some retrenchment in the forest products industry in the region. Harvest in the South continues to grow during this period, but by 2000 inventory restrictions are encountered in that region as well. As a result of limited regeneration activities on nonindustrial lands during the period from 1965 - 1985, timber of merchantable ages declines sharply in the period 2000 to 2010 and harvest falls. After 2010, however, large areas of young-growth on industrial lands in the Pacific Coast region reach merchantable size, as does timber recently planted on nonindustrial lands in the South under various public and private programs. As a result, harvest rises rapidly enough through the remainder of the projection to stabilize timber prices.

Hardwood stumpage prices are expected to grow relatively slowly over the next 15 years as hardwood inventories continue to expand. After 2000, hardwood stumpage prices increase

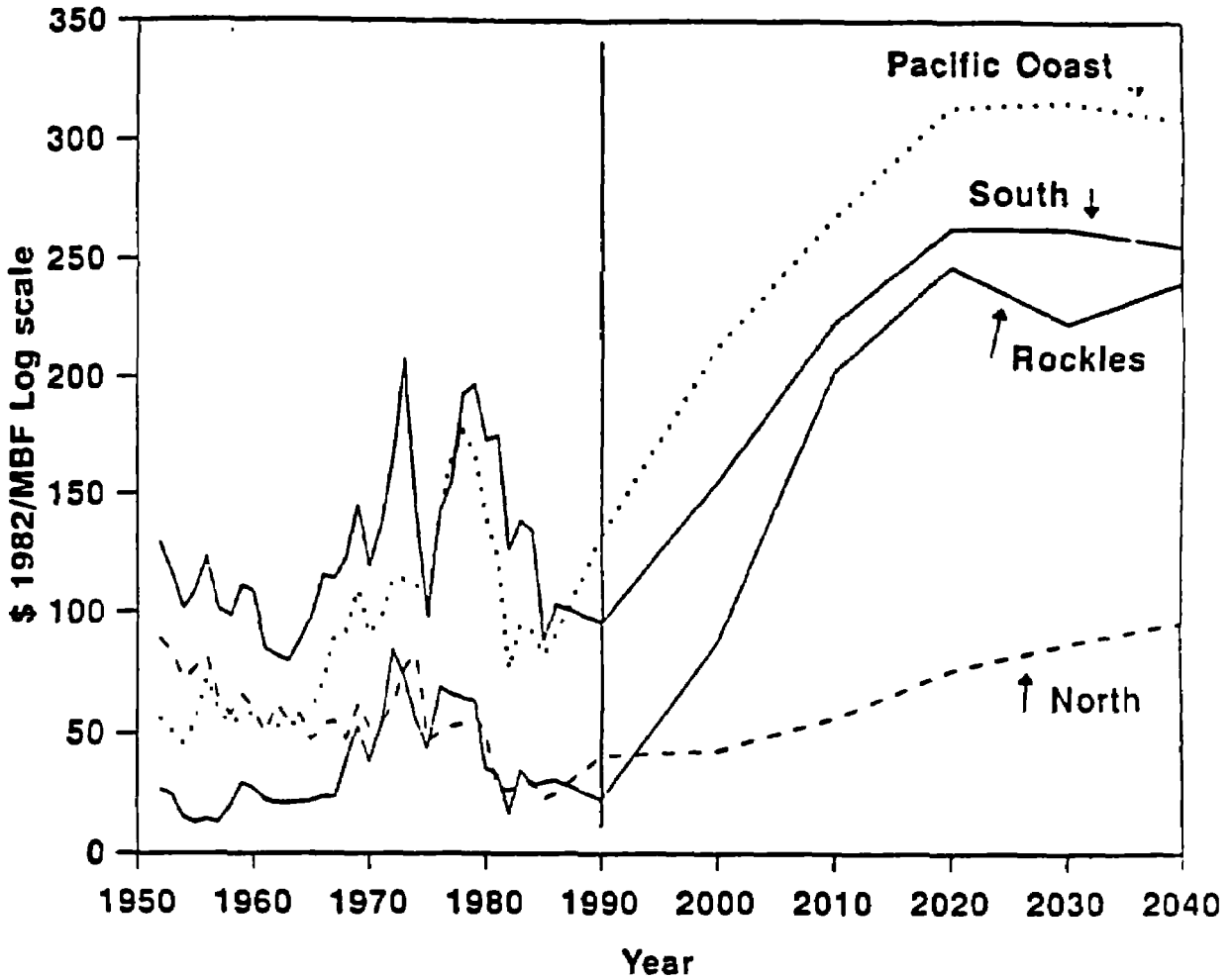

Figure 3.6. Softwood sawtimber stumpage prices from years 1952 to 1986, with projections through 1990 to 2040.

Table 3.4. Deflated price indexes for selected timber products in the US, by softwoods and hardwoods, 1952, 1962, 1970, 1976, and 1986, with projections to 2040.[1]

	Observations (previous years)					Projections (future years)				
	1952	1962	1970	1976	1986	2000	2010	2020	2030	2040
PRODUCTS. UNIT & SPECIES GROUP	Index of price per unit - 1982 = 100									
LUMBER										
Softwoods	99.8	88 3	95 3	126 0	108.0	138.6	161.8	177.5	171.8	168.3
Hardwoods	104.7	103.7	118 6	109.9	118.3	108.9	120.0	133.8	149.2	164.3
STRUCTURAL PANELS										
Plywood	172.0	119.0	109.2	143.6	109.3	120.4	136.8	152.8	148 8	154 3
OSB-waferboard				163.0	95.0	95.2	88 4	95 4	94.5	94.6
NON-STRUCTURAL PANELS										
Hardwood Plywood	184.5	174 5	153 4	110 6	90.7	88 4	86 7	85.0	83 3	81.6
Other panels[2]		151.4	115 0	92.1	107.1	107.1	107.1	107.1	107.1	107.1
PAPER & BOARD	100.2	105 3	101.5	101.5	115.8	107.6	105.7	102.9	100.3	98 9

1 Prices are measured in constant (1982) dollars and are net of inflation or deflation They measure price changes relative to the general price level and most competing materials

2 Hardboard, particleboard, and fiberboard products

3 Projections USDA Forest Service

because of declining hardwood inventories and increased demand, especially for pulpwood.

Projections of harvest by region and public/private ownerships are given in Table 3.5. Nearly all of the near-term increase in both softwood and hardwood harvest comes in the Northeast and Southeast. With only a limited basis for expanding harvest on private lands, the harvest share of the Pacific Coast region declines steadily through the projection. At the national level the ownership pattern of harvest differs between the hardwood and softwood sectors. Increased softwood harvest during the period to 2040 comes from both forest industry and nonindustrial timberlands located in the East. Increments in hardwood harvest, in contrast, come mostly from nonindustrial lands. This ownership accounted for 75% of harvest in 1986, and by 2040 the share rises to 83%. This increase reflects declining harvests from industrial lands in the East as mixed species and some hardwood sites are converted to softwood plantations.

For the US as a whole, softwood inventories remain relatively constant until 2000, then rise slowly back to their mid-1970's levels in the ensuing 40 years. Net softwood growth falls by 2000 but rises rapidly thereafter as the area of young stands increases on private lands in the West and South. Hardwood inventories, in contrast, are expected to increase in total until 2010, as losses in the South are more than offset by growth in the North and West. In later years, type conversions and loss of land to non-forest uses in the South dominates and aggregate inventory falls. Net annual growth for hardwoods, which was constant between 1976 and 1986, falls steadily through the projection period, due in large part to type conversions in the South.

Over the next 50 years the timber inventory in most regions will decline in both average size and age. For example, the average diameter of softwood timber harvested for all private timberlands in the South declines from 25.1 to 22.4 cm (9.9 to 8.8 inches) during the next five decades. The largest changes, however, are expected for softwoods on the Pacific Coast, where the average diameter of harvest falls 20% to 38.6 cm (15.2 inches). At the same time, softwood inventories on industrial timberlands in some parts of the country will be approaching something like the forester's fully regulated state, with growth and harvest in balance and a rotation near the minimum age of merchantability (roughly 25 years in the South and 45 years in the Douglas-fir subregion). These conditions are likely to exist after 2010 on industrial timberlands in the Douglas-fir subregion and the South.

Key Conclusions for Current Forest Plans scenario

Earlier assessments (e.g., USDA Forest Service 1974, 1982) have consistently projected a future with continued growth in consumption of forest products, less rapid growth in timber inventories, and rising real prices for stumpage and products. The Current Forest Plans analysis yields a similar view until about 2020, when declining growth in consumption and increasing timber inventories reduce and/or stabilize the price increases. From the Current Forest Plans case and related work we conclude that:

1. Over the next five decades, the primary source of growth in forest products consumption in the US will be in paper and paperboard. Thus,

Table 3.5. Harvest and inventory projections by region and public/private timberland owners.

Region Owner	Harvest 1986	2000	2040	Inventory 1986	2000	2040
	billion cubic feet					
Softwoods						
North						
Public	.1	.2	.2	11.3	13.9	18.8
Private	.6	1.0	1.6	36.1	42.1	41.2
South						
Public	.4	.4	.5	14.1	14.8	20.2
Private	5.3	5.6	7.5	89.7	85.4	85.6
Rockies						
Public	.5	.6	.7	81.9	83.7	89.3
Private	.3	.5	.6	18.4	18.5	12.7
Pacific coast						
Public	1.8	1.3	1.3	106.0	95.1	92.7
Private	2.1	2.3	2.9	56.3	46.2	51.1
Total US	11.2	11.8	15.4	413.8	399.7	411.6
Hardwoods						
North						
Public	.2	.3	.4	29.6	37.3	55.6
Private	1.8	3.8	4.9	110.1	136.8	145.0
South						
Public	.1	.2	.3	15.2	14.5	14.4
Private	2.8	5.3	5.3	119.0	119.8	71.5
West						
Public	.1	--	--	9.4	4.4	4.6
Private	.1	.3	.4	17.3	17.7	20.7
Total US	5.1	9.9	11.3	300.6	330.5	311.8

uncertainties in the outlook related to this sector, such as rates of wastepaper recycling and use, are particularly critical.

2. Price increases in solidwood products and sawtimber until 2010 are nearly inevitable unless there is a significant reduction in timber demand, as might be possible with higher levels of recycling. More intense forest management and increased planting will not cure short-term price increases or harvest shortfalls, though their long-term impacts can not be disputed.

3. Major plantation areas in the South (both on industrial and nonindustrial timberlands) and large areas of young growth in the Douglas-fir subregion, which are already in place, will cause slower softwood price growth after 2010.

4. The South will be the major source of any expansion in softwood supply over the next 50 years. If high planting rates in the South continue into the 1990's, as assumed in the Current Forest Plans case, flat or declining product and timber prices seem likely.

5. Hardwoods will increase in importance relative to softwoods in the total US harvest due to increased use in fiber products. In this expansion, the North has the potential to match the South in terms of contributions to incremental fiber output. Hardwood area and inventory will drop, however, if past trends in softwood plantation establishment in the South continue in the future.

6. Reductions in public timber harvest in the West will elicit little interowner substitution in western stumpage markets owing to limited merchantable private inventory, but national impacts will be reduced as a result of significant interregional substitution, including increased lumber imports from Canada. Western regions will continue to lose market share (in all products) to eastern regions because of rising relative wood costs. This trend will accelerate if public harvest falls, if higher rates of wastepaper recycling are realized (recycling favors the cost-effectiveness of the South compared to the West), or if major public programs of tree planting are undertaken on nonindustrial lands.

7. The potential impacts of higher wastepaper recycling appear to be dramatic especially in the East where harvest reductions would be concentrated. These impacts cumulate over time as lower harvest levels lead to more rapid inventory growth.

Scenario 3. Reduced National Forest Harvest

The Reduced National Forest Harvest scenario includes harvest reductions caused by elimination of harvest of old growth volumes in the Pacific Northwest, protection of spotted owl habitat in Washington, Oregon and California, protection of the red cockaded woodpecker in the South, elimination of below cost timber sales, and elimination of harvesting in existing roadless areas. Overall reduction in National Forest harvest (as growing stock) for the year 2000 was 20% below that in the Current Forest Plans scenario.

Past analyses have revealed that decreases in National Forest harvests are partly offset by changes in harvests of other owners or in other regions. In regions where there are sufficient pri-

vate timber supplies, decreases in National Forest harvest lead to higher stumpage prices that, in turn, increase timber harvests from private timberlands. For example, the National Forest harvest in the Douglas-fir subregion is reduced by 96 million cubic feet per year (see Chapter 8, in the 1989 RPA Timber Assessment; Haynes 1990). Total harvest, however, is reduced by only 40 million cubic feet by 2000, private harvests having increased by 56 million cubic feet per year. In the Douglas-fir subregion, these offsetting changes cannot be sustained after 2000 because of a worsening timber inventory situation. In other regions, such as the Rocky Mountains, the reduction in National Forest harvest is partly offset throughout the projection period.

In this type of future, declines in timber inventories are reflected in intensified competition for the available timber and higher prices for softwood stumpage prices. The prices in the Pacific Northwest, for example, are 17% above the Current Forest Plans base by 2040. Finally, softwood lumber prices are 2.4% higher in 2040 than in the 1989 Assessment projection. Because of the lumber price increases, total lumber consumption is down 1.2% and lumber imports from Canada are up 21% by 2040. The increase in lumber imports comes progressively after 2000 because domestic production is reduced as a consequence of the lower timber inventories and the associated higher prices. By 2040, domestic lumber production is 5.2% less than the 1989 RPA Assessment projection. There are different impacts among regions. The largest impacts are in the western states, particularly the Pacific Northwest with its large National Forest resources.

There are no significant impacts on the hardwood resource associated with this scenario, further illustrating the small role of National Forests in the hardwood sector.

There were no large differences in the projected timberland inventories between this scenario and the Current Forest Plans scenario, based on TAMM/ATLAS for private lands and inventory projections from the USDA Forest Service for public lands. As indicated earlier, for public lands there are also limitations in the approach to estimating future carbon flux based on the ratio of timber volume to total carbon. In this study we have therefore focused on the effects of harvest reductions on National Forest land in the region with the greatest differences, the Pacific Northwest West region.

Scenarios 4 and 5. Moderate and High Increased Recycling

These scenarios examine the impacts of two levels of additional increases in recycling in the forest sector. There is a growing interest in the increased use of wastepaper as raw material for paper and board production. Producers in other developed countries (e.g., Japan and European countries) use about twice as much wastepaper as raw material for the production of paper and board as US producers. In the US this recent interest seems to stem largely from concerns about waste disposal rather than concerns about raw material availability. In the Moderate scenario, the impacts of increasing wastepaper use to 41% of total fiber furnished were examined. The High Recycling assumption is derived from suggestions made by the American Paper Institute (API) and differs from the first primarily in how soon the higher levels of recycling are achieved. The API proposal is that recycling levels reach 45% by

1995. Projected rates of fiber recycling for the RPA and both increased recycling scenarios are shown in Table 3.6.

The Moderate Recycling Assumption was analyzed as an alternative future in the 1989 RPA Timber Assessment (Haynes 1990). Increased recycling led to a 3.7% reduction in total demand for forest products by 2040. Consumption of sawtimber and pulpwood is shown in Table 3.7. Also, by 2040 pulpwood use was 17.6% less than for the RPA scenario due to increased use of wastepaper. Some of the savings in wood that would have been used as pulp are being used for the manufacture of other products, especially lumber. US softwood lumber consumption rises 2.8%, imports of softwood lumber from Canada drop by 48.2%, and US softwood lumber production rises by 12.2%. The largest harvest reductions are for softwoods, particularly in the Pacific Coast states where harvest falls, both because of lower pulpwood use and because of competition from increased lumber and plywood production in the South. Reductions in harvest volumes in the South are larger for hardwoods than for softwoods.

Another way to look at this relationship is in terms of reductions in area harvested due to increased recycling. As the use of wastepaper increases, fewer acres are harvested each year (440,000 acres less in 2010). The bulk of this unharvested area is in the South, and is evenly split between hardwoods and softwoods. Changes in wood prices are another way to gauge the impact of increased use of wastepaper. In the South, the reductions in harvest have fairly substantial price impacts. By 2010, sawtimber stumpage prices are expected to be reduced by 19.7%. This makes the region more competitive compared to Canada, for example, and raises lumber production by 26% by 2040. Hardwood sawtimber stumpage prices, on the other hand, are reduced by only 0.5%, reflecting the relatively abundant supplies of hardwood pulpwood and the limited interaction between hardwood pulpwood and sawtimber markets.

Scenarios 6, 7, 8, 9. Afforestation

Several studies have examined the potential land available for afforestation in the US, as well as the associated costs of forest stand establishment (Moulton and Richards 1990, Parks and Hardie 1992). These studies provide the basis for estimating areas, by forest type and region, which could be planted given a specified investment. For the present study, four scenarios have been developed based on investments of $110 and $220 Million per year for 10 years applied to afforestation schemes described by Moulton and Richards (1990) and Parks and Hardie (1992). The assumption has been made that after 10 years of annual payments, the land owner will maintain the land as forest for the course of the rotation and beyond.

The primary source of land for afforestation was marginal pasturelands, the majority occurring in the South Central region (Table 3.8). Allocations among regions, and among forest types within regions, were made based on the distributions in Moulton and Richards (1990) and Parks and Hardie (1992). The yield tables associated with each forest type, which are required by the ATLAS forest inventory model, were likewise derived from the yield estimates in the original documents. Stand level carbon budgets were built as described in Section 3.A with two exceptions. First was the absence of a carry over in the woody

Table 3.6. Projected percent of total fiber from recycled wastepaper in RPA, Moderate Recycling, and High Recycling scenarios.

Year	RPA Assumption	Moderate Recycling Assumption	High Recycling Assumption
1986	20.9	.209	.209
2000	21.0	.285	.450
2040	28.0	.415	.450

Table 3.7. Projected consumption of sawlogs and pulpwood in RPA and Moderate Recycling scenarios.

	Sawlogs		Pulpwood	
Year	RPA	Moderate Recycling	RPA	Moderate Recycling
	billion cubic feet			
1986	9.0	9.0	5.8	5.8
2000	8.2	8.3	7.4	7.1
2020	9.5	9.6	9.6	8.4
2040	9.6	9.9	10.8	8.9

debris and forest floor pools, since there was no previous accumulation, nor inputs from debris associated with timber harvest. Second was an allowance for increases in the soil carbon pool during stand development.

The basis for the accumulation of soil carbon is field studies suggesting significant reductions in soil carbon when land use changes from native vegetation to agricultural production or grazing (e.g., Mann 1986), and a reversal of that process in cases of reforestation (e.g., Jenkinson 1971). The reduction is driven by reduced inputs of organic residues, and increased rates of decomposition, associated with greater aeration and higher temperature of the soil. Typically, losses are on the order of 10-50% in the temperate zone (Schlesinger 1986, Johnson 1992a, 1992b). The soil carbon accumulation after reforestation is associated with humic materials derived from decomposition of leaf litter and dead roots. Such gains are on the order of 30-50 g m^{-2} yr^{-1} in the temperate zone (Jenkinson 1971).

For this analysis, we have set initial soil carbon for the planted areas at 20% below the average for the relevant forest type and region (Section 3.A). Since most of the converted land is marginal pastureland, this intermediate value

Table 3.8. Enrollment schedules of forestation of marginal lands by region for two funding levels and two studies.

TAMM/ATLAS Region	Funding = $110 M per year for 10 years		Funding = $220 M per year for 10 years	
	Parks/Hardie ha x 10^3	Moulton/Richards ha x 10^3	Parks/Hardie ha x 10^3	Moulton/Richards ha x 10^3
SC	2607	2335	3453	3257
SE	0	0	153	0
NC	0	0	998	0
NE	33	0	94	0
RM	0	0	0	736
PSW	0	0	26	0
PNW	192	0	244	0
TOTAL	2832	2335	4968	3993

was deemed appropriate. Converted croplands would start with a lower soil carbon level. Soil carbon was then assumed to return to the average level over the course of one rotation. These assumptions result in soil carbon accumulation rates ranging from 30-50 g m^{-2} yr^{-1} in the southern regions and 15-25 g m^{-2} yr^{-1} elsewhere.

The accumulated wood volumes on the afforested lands were permitted to be harvested within the TAMM modeling framework. Harvested areas were assumed to be replanted in the same forest type and assigned a yield table associated with high productivity but low management intensity.

Scenario 10, 11, 12. Combinations

The individual forest policy scenarios previously described include effects that tend to either reduce supply of forest products (e.g., reductions in National Forest harvest levels) or increase supply (e.g., recycling and afforestation). In order to evaluate the potential for combining these options in ways that minimize price impacts and avoid sharp changes in social welfare, an additional suite of combination scenarios was examined.

The first combination scenario paired the Reduced National Forest Harvest scenario with the assumptions of Moderate Recycling. The blend of these scenarios was anticipated to have opposing effects on timber supply, thus dampening price fluctuations. The second combination included all the assumptions of the first and added a doubling of timber exports. A third scenario combined the $110 million per year afforestation scenario of Moultan and Richards with the conditions of the second scenario.

SECTION 4. RESULTS AND DISCUSSION

A. Current pools

Over half of the total timberland carbon is in the mineral soil (Figure 4.1). Tree carbon, which includes coarse roots, is the next largest component at 31%, followed by woody debris (11%), forest floor (6%), and understory (1%). Total carbon storage in living trees on timberland in the US was estimated at 11.1 Pg-C. This quantity conforms with the estimate identified in Box 6, which is based on biomass factors derived from Cost et al. (1990) and Koch (1989). The comparable estimates of Birdsey (1992a) was 11.4 Pg-C. The addition of woodlands carbon, as described in Section 9.B, brings the total for all forests in the US to 11.6 Pg-C.

Privately-owned forests contain 64% of timberland carbon. In terms of the regional distribution, the Northeast and South Central regions have the largest absolute quantities of carbon (Figure 4.2). A different pattern is seen for the average quantity of carbon per unit area, with the highest average found in the Pacific Northwest West and the lowest in the South Central region (Figure 4.3).

The relatively large pool of woody debris demonstrates that consideration of this pool is necessary for a comprehensive analysis of forest carbon. Recently harvested stands may have large quantities of woody debris in the form of dead roots, stumps, and logging residue. Relatively old stands contain large quantities of woody debris because of the accumulation of snags and fallen logs. Because of the limited availability of inventory data on woody debris, we have relied on a modeling approach in the present study to estimate woody debris pools (Section 3.A.1). Better inventory data on variability in the woody debris pool among forest types and age classes is needed and would allow for more refined analyses.

The uncertainties in the pool estimates for private timberlands have been discussed (Section 3.A, 9.A) and relate in part to the intensity and frequency of sampling by the FIA surveys. For public lands, we have depended on area and volume inventories, primarily provided by the USDA Forest Service, which vary widely in their reliability. The relatively limited availability of data regarding forest stand age class distributions on public lands suggests the need for additional study. Satellite remote sensing holds considerable promise for both vegetation classification and evaluation of forest structure (Iverson et al. 1989; Loveland et al. 1991; Turner et al., in press). Research efforts in this area are underway at the US EPA Environmental Research Laboratory, Corvallis, Oregon, and may provide new evaluations of forest condition on public lands in the western US as well as verification of USDA Forest Service inventory data.

B. Current flux

1. Base year biologically driven carbon flux

The estimates of carbon flux derived from the stand-level carbon budgets indicate a common sequence over the time course of the analysis (Figure 4.4).

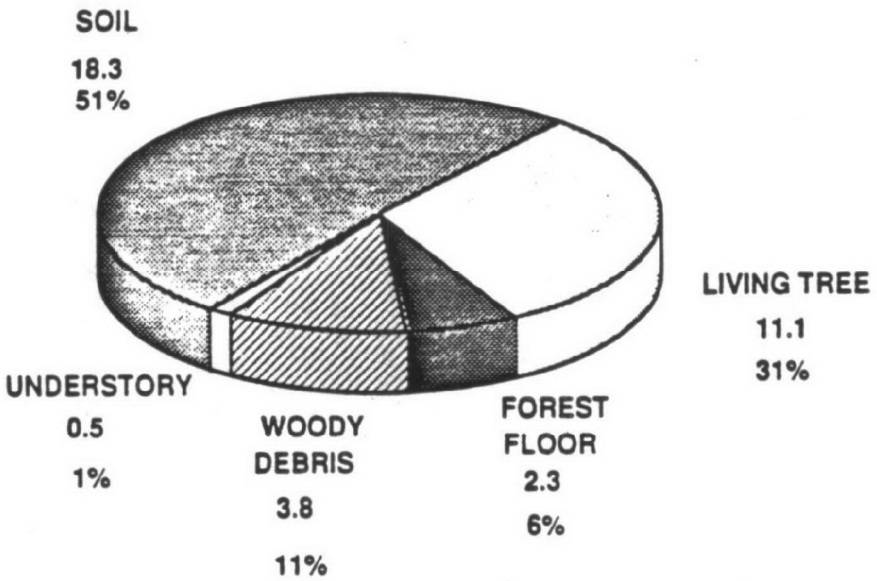

Figure 4.1. Relative contribution of carbon pools (Pg-C) to total carbon for US timberland.

Early in stand development the net ecosystem productivity (NEP), or net change in the total carbon pool, is negative because of carbon loss to the atmosphere associated with decomposition of the woody debris pool. Near the time of canopy closure, the rate of carbon accumulation associated with tree growth begins 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. Also, microbial respiration from decay of woody debris increases in old stands. The time course of the fluctuations in NEP varies with forest type (Figure 4.4). NEP in the Southeast planted pine forest type peaks much earlier than in the Pacific Northwest West Douglas-fir and Northeast Maple-Beech-Birch forest types.

The overall net flux of carbon driven by biological processes was 286 Tg-C yr^{-1} moving from the atmosphere into forest stands. Private timberlands accounted for 78% of the total US uptake. The two southern regions had the highest absolute uptake (Figure 4.5) followed by the two northern and four western regions. The net uptake per unit area (Figure 4.6) was also highest in the South East and South Central regions (0.18 to 0.21 kg-C m^{-2} yr^{-1}), followed by the Pacific Northwest West, the Northeast, and the Pacific Southwest (0.14 kg-C m^{-2} yr^{-1}).

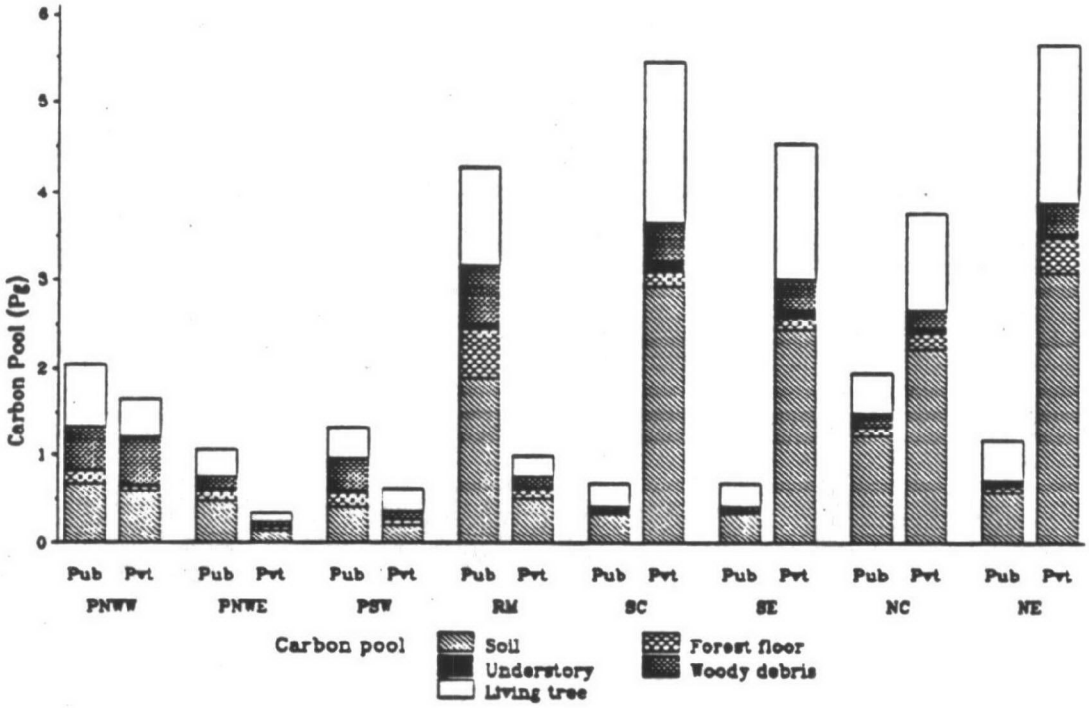

Figure 4.2. Base year carbon pools (Pg-C) by region, ownership, and component for US timberland. Region abbreviations are same as in Figure 3.2. Pub = public ownership, Pvt = private ownership.

Box 6. Literature-Based Estimate of Timberland and Woodland Biomass Pools

The Forest Carbon Model (FCM) uses inventory data in combination with a set of stand level carbon budgets to estimate current carbon reservoirs. As an effort to validate the estimate for biomass, an analysis was made of the current biomass in timberland and woodlands using existing literature data.

Cost et al. (1990) was the primary source for estimates of aboveground wood, bark and foliage on commercial timberlands. In that study, FIA plots, Bureau of Land Management inventory, and stand volume information from National Forest plots were all used for volume estimates. These estimates were then converted to biomass by Cost et al. using existing information from the literature, the extent of which varied by region (there tends to be much more detailed county by county information and biomass information in the East than in the West). This approach estimated above-ground biomass of live sound and cull trees, including tops, foliage, and biomass of seedlings and saplings. In their report, Cost et al. give ovendry weight, which has the advantage over green weight in that the latter contains inherently larger differences between softwoods and hardwoods (water content affecting specific gravity) and even among species within these fiber type categories. An example of detailed state/county information which is available for some Eastern states can be found for Alabama (FIA Staff 1985).

Belowground estimates for biomass were adapted from Koch (1989) as a function of the ratio of aboveground to belowground biomass.

Region	Softwood $\%T_B$	Hardwood $\%T_B$
Western	14.6%	15.5%
Eastern		
North	17.0%	15.5%
South	16.3%	19.7%

$\%T_B$=percent root biomass of total tree biomass, summarized from Koch (1989).

Koch's estimates are for trees > 12.7 centimeter (5 inches). The above averages exclude white spruce and alpine fir, where $\%T_B$ was approximately 21%. For both timberlands and woodlands, total biomass for unreserved or available lands on a per hectare basis was applied to reserved area for which no biomass estimates are available.

Softwood and hardwood biomass estimates were converted to carbon using the conversion factors (0.491 for hardwoods and 0.521 for softwoods) in Koch (1989).

Woodlands

Since biomass estimates from Cost et al. (1990) were limited to commercial unreserved timberlands, volume and biomass for woodlands were derived from current resource area and state inventory reports. Major emphasis was placed on the Western US, because the vast proportion of woodlands are in the West. The data requested from FIA groups for woodlands by forest type within state, included reserved and unreserved area, total volume, dead volume, and number of trees by diameter class. Whenever possible, the diameter class information was used in conjunction with equations to estimate top and branch weights. Much of the data came from volume and growth analyses in progress, with a strong effort by FIA to provide us with at least preliminary results in some cases or advice as to applying average growth data to woodland area for which biomass information has not been summarized.

The average kg ha^{-1} of biomass compiled for the West was then assumed to hold for the relatively small area classified as "Woodland in the East." References corresponding to biomass estimates for given states and forest types or species are given in Section 9.B. All data used are on file with the US EPA Environmental Research Laboratory in Corvallis, Oregon.

Estimates of Tree Carbon Pools

The estimates for tree carbon pools are summarized by region in the table below. We estimated aboveground carbon storage of US forests to be 10 Pg-C, of which 9.6 Pg-C is in timberlands, with woodlands accounting for the remainder (Section 9.B). The literature-based estimate, exclusively for the tree carbon in forests in the conterminous 48 states and including roots, is 11.6 Pg-C.

Tree Carbon Pools for Forests in the Conterminous US[1]

Region[2]	Forest Area (Mha)			Tree Carbon Pool[3] (Pg-C)[4]			
				Timberland		Woodland	Total
	Timberland	Woodland	Total	Above	Below		
West Coast	25.6	10.6	36.2	1.63	0.25	0.12	2.00
Intermountain	28.2	27.8	55.9	0.97	0.15	0.32	1.44
North Central	32.9	1.4	34.2	1.31	0.20	0.02	1.53
Northeast	34.0	0.5	34.5	1.73	0.29	0.01	2.02
Southeast	35.1	0.5	35.5	1.75	0.26	0.01	2.03
South Central	45.3	1.6	46.9	2.23	0.36	0.02	2.60
Total	201.1	42.4	243.2	9.62	1.50	0.50	11.62

1 See text for data sources

2 West comprises Washington, Oregon, and California, North Central includes Great Plains states, remainder follow Waddell et al 1989

3 Includes estimates for saplings, stumps and foliage on timberlands

4 1 Petagram (Pg-C) = 1 billion metric tons

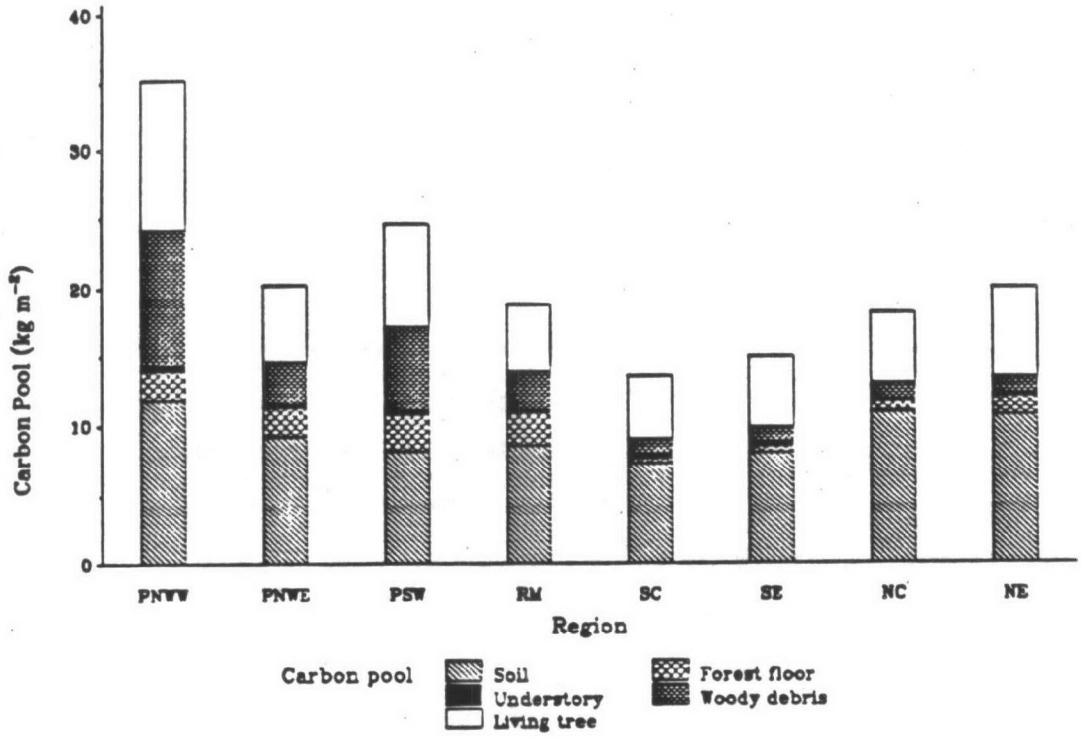

Figure 4.3. Base year carbon storage (kg-C m^{-2}) per unit area by region and component. Region abbreviations are the same as in Figure 3.2.

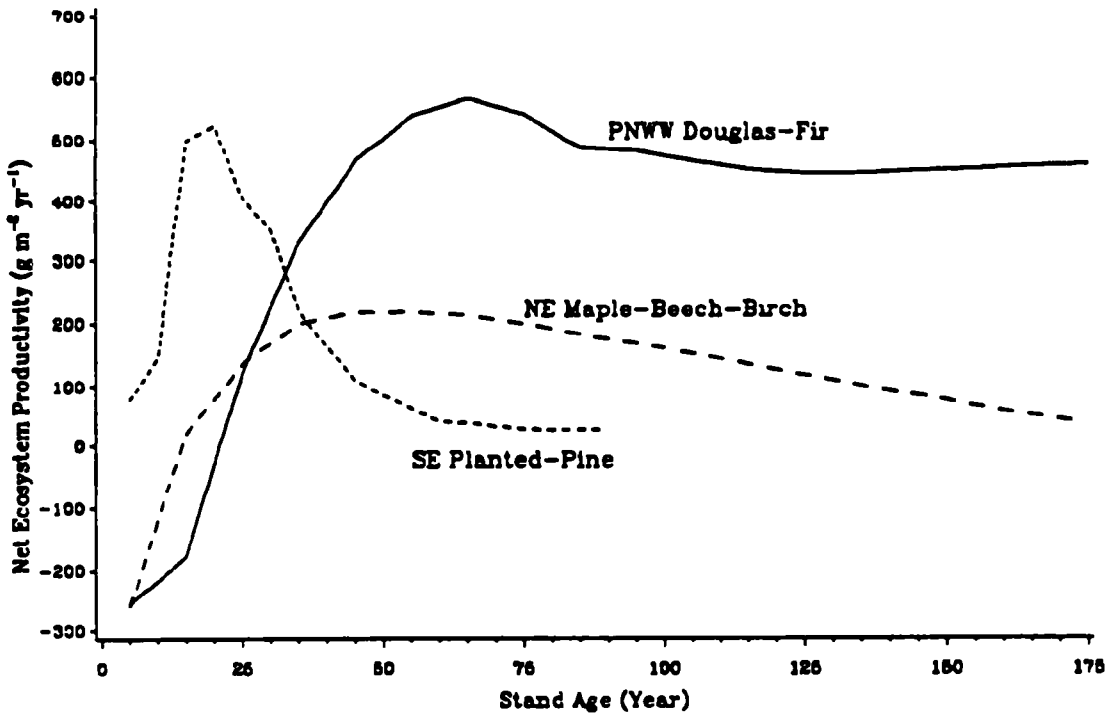

Figure 4.4. Net ecosystem productivity (g-C m^{-2} yr^{-1}) of selected forest types. Positive values are carbon transfer from the atmosphere to the land base. Negative values are carbon transfer to the atmosphere. Region abbreviations are the same as in Figure 3.2.

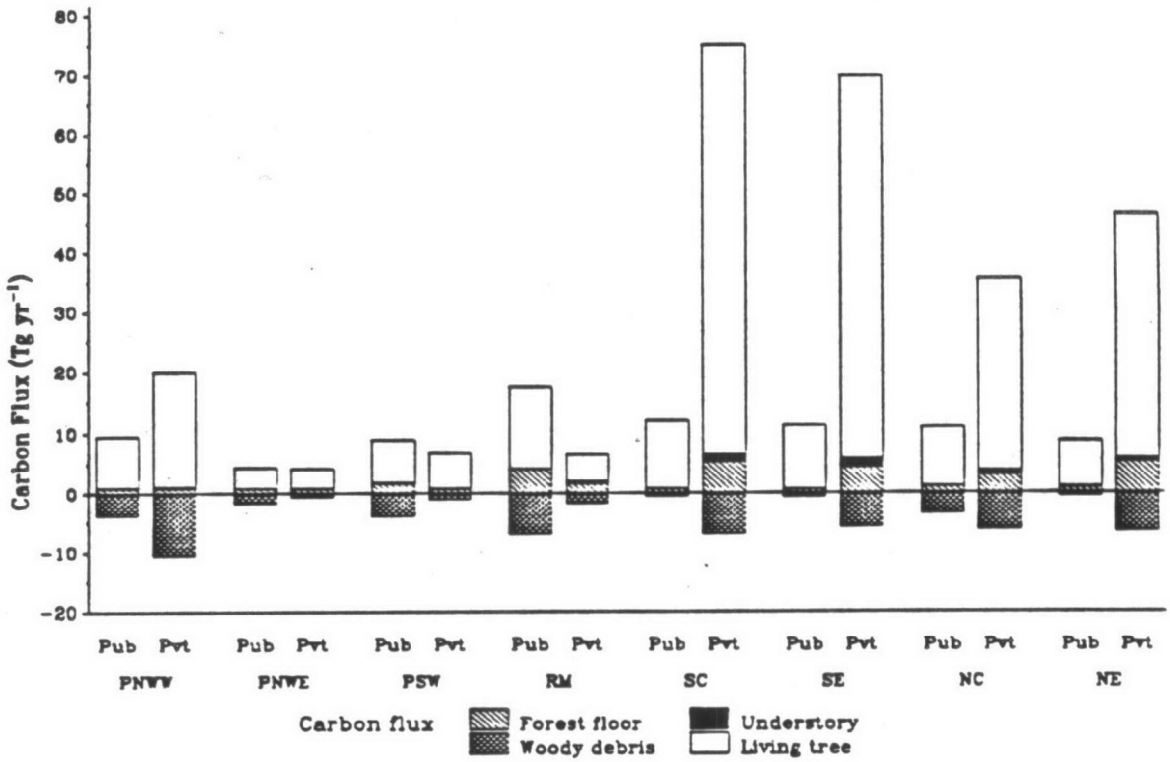

Figure 4.5. Base year biologically-driven carbon flux (Tg-C yr^{-1}) by region, ownership, and component for US timberland. Positive values are carbon transfer from the atmosphere to the land base. Negative values are carbon transfer to the atmosphere from the land base. Region abbreviations are the same as in Figure 3.2. Pub = public ownership, Pvt = private ownership.

The Rocky Mountains had the lowest uptake per unit area (0.05 kg-C m^{-2} yr^{-1}), reflecting relatively dry and colder climates.

In the Pacific Northwest West, where the age class distribution on public lands was taken into consideration, private lands accounted for 62% of the total uptake but only 45% of the total timberland area. That difference 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 greater than 150 years of age, while the comparable value for Douglas-fir stands on forest industry lands (based on the RPA database) was about 5%.

The only pool to consistently show a net carbon emission was the woody debris pool. This pattern is driven by the decay of woody debris in the early to mid stages of stand development. During this period, the quantities 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 compartment due to tree mortality. Woody debris decomposition provided the greatest offset to tree growth in the Pacific Northwest West and had a minor impact in the eastern regions (Figure 4.6).

Limited opportunities exist for validation of the base year flux estimates. Waddell et al. (1989) estimate timber volume growth on US available timberland at 22.3 x 10^9 ft^3 for 1986. The comparable value in the present analysis is 21.0 x 10^9 ft^3, a difference of 6%. The difference in the growth estimate occurs predominately on public lands where there is the greatest uncertainty about inventories and growth rates.

2. Base year harvest-driven carbon flux

The national growing stock reduction associated with harvest amounted to 130 Tg-C yr^{-1} for the base year, with the largest quantities occurring in the South Central and Southeast regions (Figure 4.7). To convert growing stock reduction data to actual carbon removal from forests, several factors must be considered. Not all growing stock reduction results in carbon removal--portions are left on site as logging residues. But in addition to growing stock, other biomass (non-growing stock) is removed in the form of tree tops, boles of trees less than 12.5 cm dbh, branches and other material. In the past decade, considerable quantities of hardwood have been salvaged from US forests for use as fuelwood (Haynes 1990). Using the ratios of growing stock reductions to total removals by Haynes (1990), total removal was estimated to be 139 Tg-C yr^{-1}.

As noted in Section 3.B, the effect of timber harvest is manifested in several ways. Most significant is the transfer of a large quantity of carbon out of the tree biomass pool. In the US, this removal is a large proportion of the annual carbon accumulation from tree growth. The difference between annual growth and harvest removals is greatest in the Northeast, Southeast, and Southcentral regions.

Besides the direct removal of carbon by harvesting, there is the transfer of carbon from the living tree pool into the woody debris pool. The difference between total tree carbon and merchantable wood carbon, for the harvested area, should show up in the woody debris pool. However, comparison of the woody debris pool in the

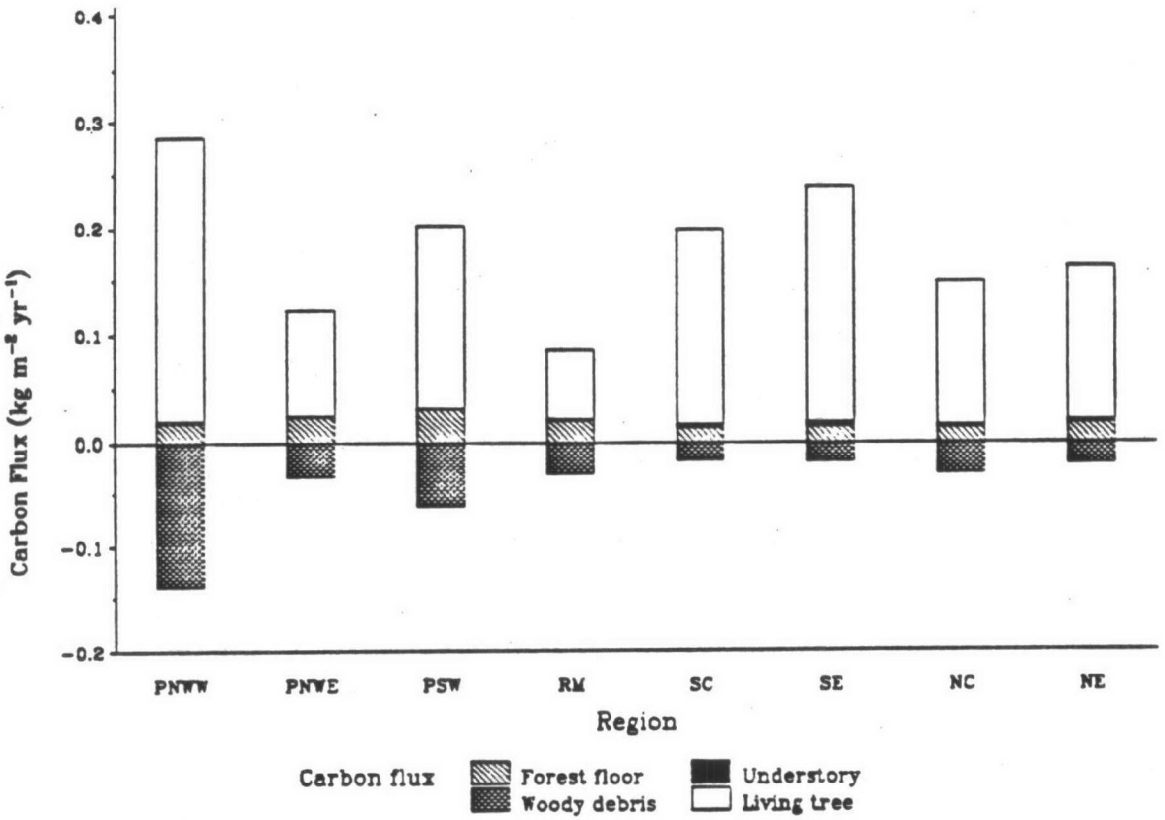

Figure 4.6. Base year biologically-driven carbon flux (kg-C m^{-2} yr^{-1}) per unit area by region and component. Positive values are carbon transfer from the atmosphere to the land base. Negative values are carbon transfer to the atmosphere from the land base. Region abbreviations are the same as in Figure 3.2. Values are the sum of public and private ownership.

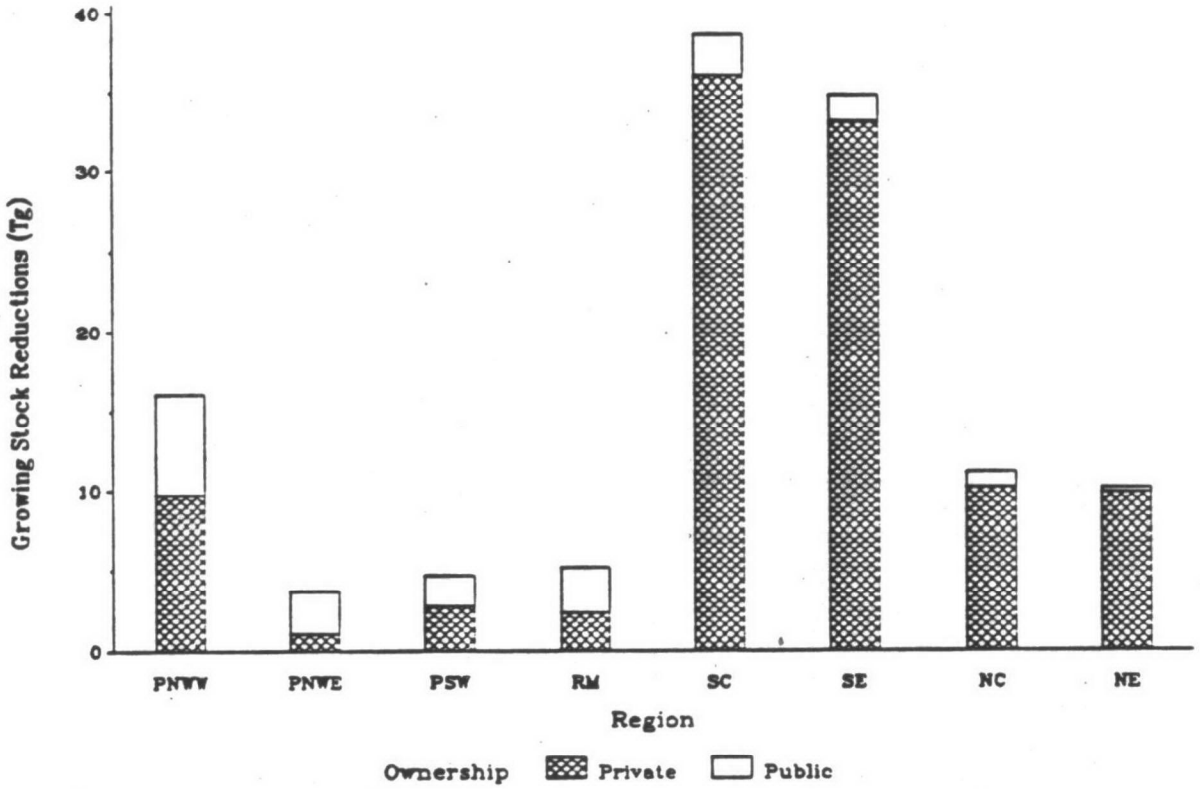

Figure 4.7. Base year growing stock reductions due to harvesting (Tg-C) by ownership and component. Region abbreviations are the same as in Figure 3.2.

Table 4.1. Base year timberland carbon budget. Positive values (Tg-C) are net accumulations and negative values are net reductions.

	Biological	Harvest	Combined
Tree	310	-248[1]	52
Woody Debris	-60	96	36
Forest Floor	31	-26	5
Understory	5	-7	-2
TOTAL			91

[1] Includes harvest removals (139), woody debris formation (96) and loss to land use change (13).

youngest age class and at rotation age, for the area harvested and replanted, did not account for all of it. Part of this deficit is related to loss of woody debris associated with landuse change. The fate of this carbon is unclear but it is needed to balance the woody debris budget in the base year analysis. Part of the deficit in woody debris accumulation may also be a function of an underestimate of the woody debris pool in the youngest age classes. If so, then inputs to the woody debris pool may be greater than is simulated by the stand level carbon budgets (Section 3.A).

The rapid change in forest floor and understory carbon associated with or immediately following harvesting amounted to a 33.0 Tg-C yr^{-1} loss of carbon. Total area harvested was about 2.85×10^6 ha.

3. Complete base year flux analysis

Considering both the biological and harvest-related transfers, the net carbon accumulation on the land base was 91 Tg-C (Table 4.1). The majority of the carbon accumulation was in the tree (52 Tg-C) and woody debris (36 Tg-C) pools. The forest floor showed a small gain (5 Tg-C) and the understory a small loss (-2 Tg-C).

If the harvested carbon entered into permanent storage, the forest product pool would constitute an additional sink of 139 Tg-C yr^{-1} for the base year. But determination of the net carbon transfer requires consideration of carbon emissions associated with the decay or burning of these and previously produced forest products. Based on the analysis in Box 8, we estimate a net change in the products pool of only 36 Tg-C yr^{-1}.

Annual carbon emissions from wildfire on US forests vary widely between years. Based on examination of USDA Forest Service statistics on fire frequency and extent over the last several decades (A. Auclair, Science & Policy Associates, personal communication) an estimate of 20 Tg-C was used in the present analysis. Approximately one-third of the total emissions are associated with woodlands, which tends to maintain a long-term carbon equilibrium on those lands. Direct emissions from burning of timberlands were thus estimated at 13 Tg-C.

Box 7. Net Accumulation in Product and Landfill Pools

Unlike the forests, there are no systematic inventories of the product or landfill carbon pools. Neither the total amount of carbon in these pools, nor the distribution among constituent pools, is known. The model HARVCARB (Row and Phelps 1990; Section 9.E of this document) can be used to estimate the flux to products and landfills from newly harvested biomass. However, because the initial pool sizes are not known, HARVCARB cannot calculate flux from existing stocks of forest products in use or in landfills. Thus, to obtain an estimate of the net flux associated with the product and landfill pools, it was necessary to use published national values for the production of forest products, for the transport of forest products to landfills, and for the release of carbon from landfills. Based on analyses described here, the net flux of carbon into the product and landfill pools was estimated to be 36 Tg-C yr^{-1} in 1990 (20 Tg-C yr^{-1} stored as products, 16 Tg-C yr^{-1} accumulated in landfills).

Product pool. Paper production in the US was 85.2 Tg in 1990. Of this, 63.7 Tg was from US sources of pulp and 21.5 Tg from imported sources of pulp (US DoC 1991). In the same year, 209 Tg of waste were transported to landfills. Paper and paperboard account for 41% of waste transported to landfills, or 85.7 Tg (EPA 1991). Thus, the pool of carbon in paper is approximately in steady state; that is, the net flux is close to zero.

In 1990, 21.8 million m^3 (9.2 billion board-feet) of hardwood lumber was produced in the US. At a density of 392 kg m^{-3} (VanHooser and Chojnacky 1983), hardwood production was 8.5 Tg. Softwood lumber consumption was 106 million m^3 (45.0 billion board-feet), which included 78 million m^3 (32.9 billion board-feet) produced in the US and 28 million m^3 (12.1 billion board-feet) imported into the US (US DoC 1991). Using a density of 498 kg m^{-3} for softwood lumber (Van Hooser and Chojnacky 1983), 38.6 Tg of softwood lumber was produced from US sources and 14.2 Tg was imported. Thus, total production of lumber from US sources was 47.1 Tg out of a total consumption of 61.3 Tg. Wood accounts for 3.7% of the annual waste stream to landfills or 7.7 Tg (EPA 1991). If the fraction of US-produced wood is the same in the waste stream as it is in the production-stream, then 5.9 Tg of US-produced wood was transported to landfills. Thus, the US-produced wood-product pool increased by 41.2 Tg. Assuming 50% of this was carbon, then the US-produced wood-product carbon pool increased by about 20 Tg in 1990. This is an overestimate because there are losses from the product pool in addition to transport to landfills.

Landfill pool. A total of 93.4 Tg of paper and wood products (85.7 + 7.7 Tg) was transported to landfills in 1990. Assuming 50% carbon content, the amount of carbon in this flux was about 47 Tg. Emission of methane (CH_4) from US landfills is about 18 Tg yr^{-1} (EPA 1991). Because methane is 75% carbon, this flux contains 13.5 Tg of carbon. Assuming that an equal amount of CO_2-C was emitted (EPA 1991), total carbon flux from US landfills is about 27 Tg yr^{-1}. If all of the emitted carbon came from decomposition of forest products, then the pool of forest products in landfills increased by about 20 Tg of carbon in 1990. Considering that 75.6% of paper and wood products are produced from US sources, then the increase in wood products in landfills from US sources was about 16 Tg. This is clearly an underestimate because some of the emitted carbon came from other sources, such as food and yard wastes.

Summation of the carbon flux processes described here suggests that the forest sector in the conterminous US was a sink for about 114 (91+36-13) Tg-C yr^{-1} of carbon for the base year 1990. By way of comparison, carbon dioxide emissions (as carbon) from fossil fuel and cement manufacture in the 1980's averaged about 1,300 Tg-C yr^{-1} (Boden et al. 1992). Thus the forest sector offset about 8% of US carbon emissions in 1990.

Some of the uncertainties regarding specific components of the carbon pools and flux analysis have been discussed earlier. Considering the general agreement that forest growing stock volume is increasing in the US (Waddell et al. 1989, Birdsey 1992a), it seems unlikely that the sign of the estimated net flux is wrong, i.e. US forests are currently a net sink for carbon. The estimate for the absolute magnitude of the current sink is less certain. Alternative assumptions about post-harvest losses and gains in soil carbon (Birdsey 1992c) would probably increase the estimated sink. Using the net growth of growing stock on public lands in Waddell et al. (1989) would also increase the sink. An outline of future research aimed at reducing the uncertainties in the present analysis is given in Section 6.

C. Forest policy scenarios

During the period 1980-1988, US emissions from fossil fuel use increased annually at a rate of 6.5 Tg-C yr^{-1}, or about 6% of the sink-strength of the forest sector. Increase in emission rates were greater than this toward the end of the period because of increased economic activity. A sustained increase in the sink-strength of the forest sector could offset some or all of the expected increase in emissions from fossil fuel use. In this section, we describe several forest policy scenarios and their projected influence on the forest sector carbon budget.

1. The 1989 RPA Assessment scenario

Private Lands

The total non-soil forest carbon budget is impacted over time by both changes in the land base and changes in the age class distribution of the forests and, to a lesser extent, by modification of forest management techniques. For private timberland, the trend in total non-soil carbon storage is upward in the 1990's. Thus the net accumulation for private lands is a carbon storage increase of 46 Tg-C yr^{-1} for the period 1990-2000 (Figure 4.8). After 2010, total carbon on private forest decreases. Some of the carbon lost from private forests will be emitted to the atmosphere and some will be found in products and landfills.

The fact that over the course of the 50-year scenario the acreage of private land forests is reduced 7% (Figure 4.9) accounts for some of this carbon loss but is not the dominant factor. The more important factors are the continuing increase in the harvest level (Figure 4.10) and a trend towards more intensively managed forests which are characterized by younger average age classes and lower average carbon storage (e.g., Cooper 1983, Harmon et al. 1990).

Early in the scenario, increases in the live tree carbon component of forests, particularly in the Northeast region, more than offset the loss of carbon due to land use change and management intensification. Private land in the Pacific Northwest West also accumulates carbon in the 1990's as the age

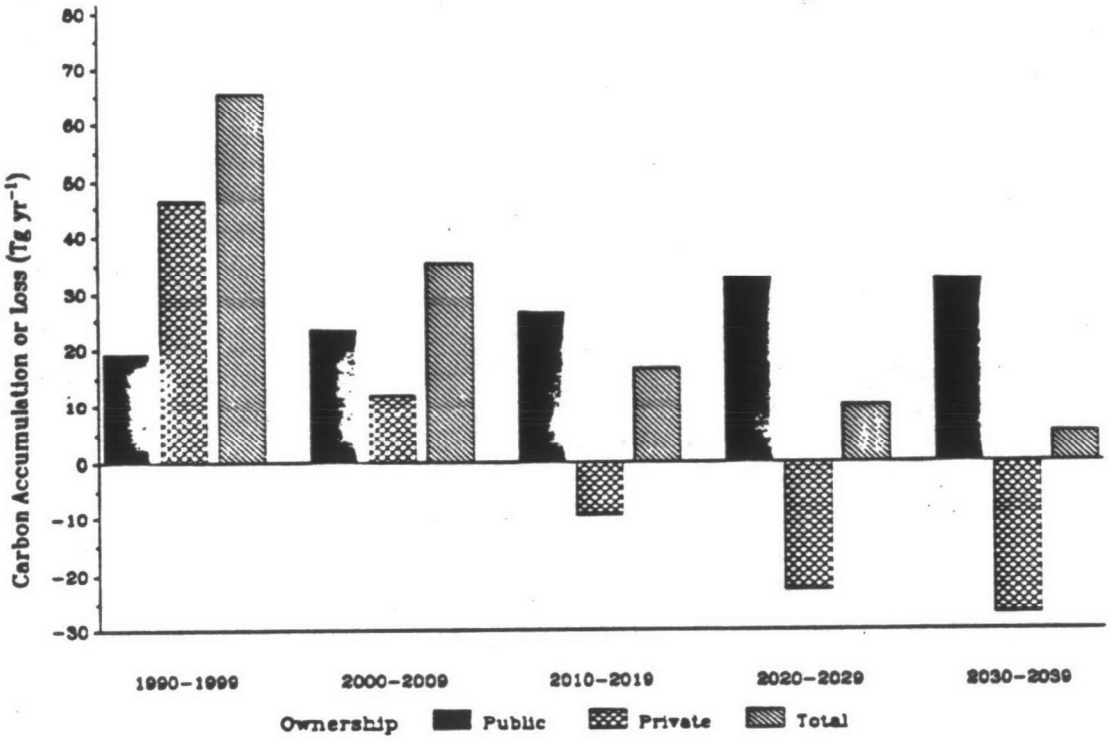

Figure 4.8. Projections of net carbon accumulation or loss (Tg-C yr^{-1}) for US timberlands (sum of available and reserved) for the RPA Assessment scenario. Positive values are net carbon accumulations. Negative values are carbon losses.

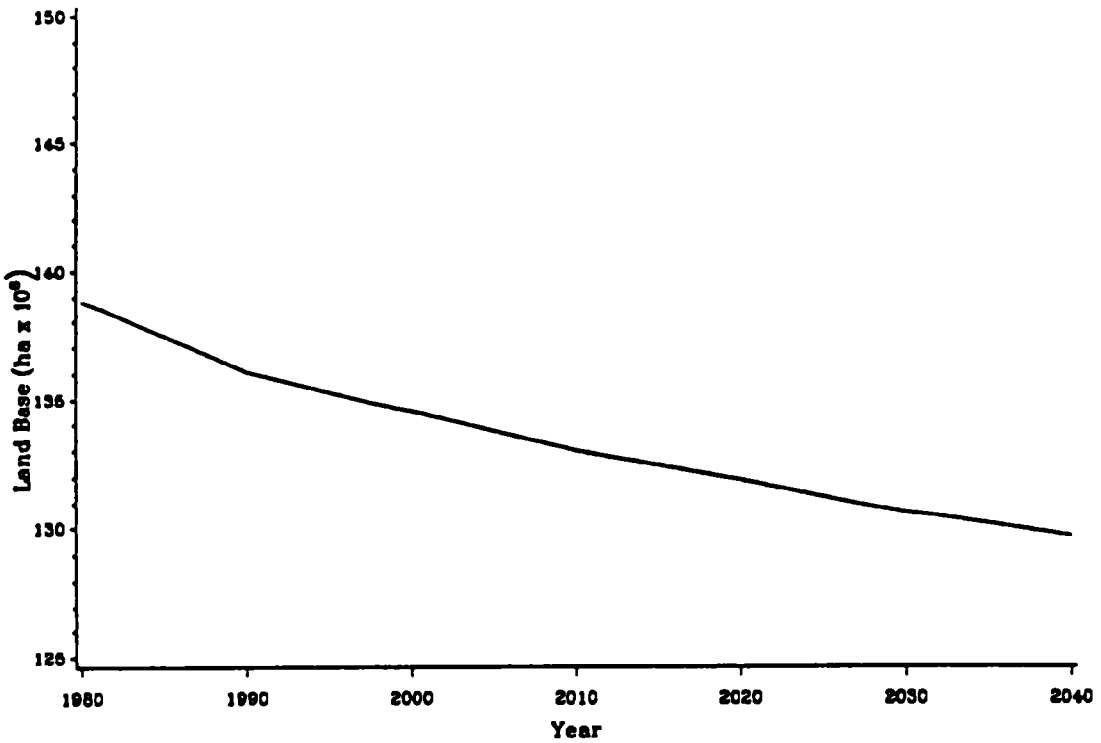

Figure 4.9. Forest land base (ha x 10^6) for private timberlands.

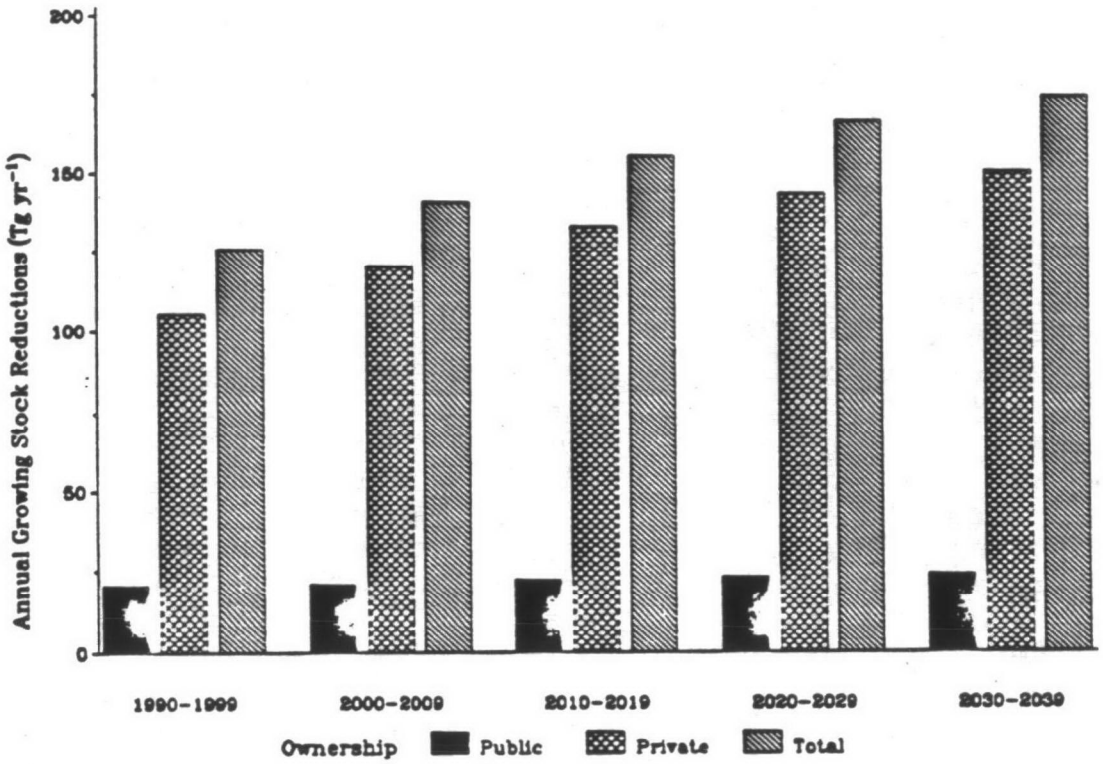

Figure 4.10. Projected annual growing stock volume reductions (Tg-C yr^{-1}) from US timberlands for the RPA Assessment scenario.

class distribution shifts to slightly older trees. After the year 2000, total non-soil carbon begins to drop, however, affected by both decreases in the land base and decline in average tree carbon per unit area. These trends are discussed further in Haynes (1990).

Public lands

In contrast to private holdings, the forest acreage on public lands does not change substantially over time in the RPA Assessment scenario. On lands subject to timber harvest, carbon accumulates at the rate of 9 to 23 Tg-C yr^{-1} throughout the 50-year scenario for the nation as a whole (Figure 4.8). Carbon reductions due to harvesting never exceed annual carbon accretion by growth. However, in the Pacific Northwest West, Pacific Northwest East and Pacific Southwest, carbon inventories decline in the first few decades (Figure 4.11). These trends reflect a policy of prioritizing harvest of high-volume old growth stands and including this acreage in future timber production rotation management (Haynes 1990).

Since harvesting is disallowed on reserved public lands, carbon accumulates at a much faster rate per unit area than on lands available for timber harvest. Stands greater than 175 years of age (90 years in southern regions) do not change in volume, due to limitations of the FCM stand-level carbon budget data. The accumulation of carbon thus tends to be underestimated. That tendency is balanced to some extent by ignoring disturbances, such as catastrophic wildfire, which would create young stands. The net accumulation for all public reserved lands was about 9 to 11 Tg-C yr^{-1} throughout the scenario. The dominant regions were the Northeast and Rocky Mountains.

National Trends

The forest land carbon accumulation rate for the conterminous US is about 65 Tg-C yr^{-1} in the 1990's (Figure 4.8). Despite continuing carbon accumulation on available and reserved public lands, the magnitude of the carbon accumulation rate declines continuously. By the decade of the 2030's, the rate is predicted to decline to 5 Tg-C yr^{-1}. A long term carbon accumulation would, in principle, not be expected on reserved lands because natural disturbances insure a mix of early and late successional stages. Carbon storage on lands managed for timber production would likewise not be expected to increase because of continuous harvest removals and cycling through stand development.

Growing stock reductions driven by harvesting increase from 130 Tg-C yr^{-1} in 1990-1999 to about 175 Tg-C yr^{-1} in 2030-2039 (Figure 4.10), with private lands accounting for most of the change. This more intensive harvest regime contributes to a declining ratio of growth to removals.

The annual carbon accumulations above do not include the possible change in carbon contained in forest products still in use or in landfills. Assuming the same ratio of carbon accumulation in forest products and landfills to total removals as was estimated for the base year (23%), this sink grows from 36 Tg-C yr^{-1} to 50 Tg-C yr^{-1} during the 50-year period.

As with the base year analysis, it should be noted that the current approach, which simply takes the difference in woody debris pools at two points in time for all forested lands, does not account for decomposition of woody debris on deforested lands. The same limitations

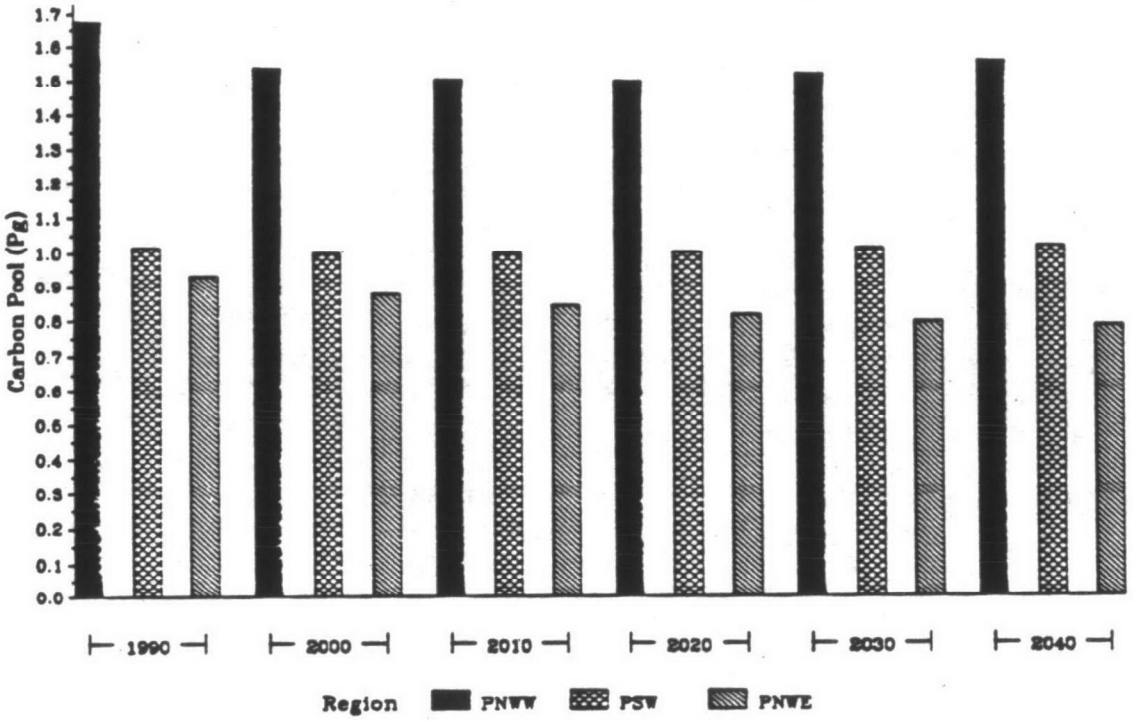

Figure 4.11. Projections of carbon pools (Pg-C) on available public lands for the PNWW, PNWE and PSW regions for the RPA Assessment scenario. Region abbreviations are the same as in Figure 3.2.

as found in the base year analysis in accounting for all woody debris from harvest also apply. An effort will be made in future analyses to carry previously forested lands in the inventory in order to account for that CO_2 source.

2. The Current Forest Plans scenario

The primary effect of the Current National Forest Plans scenario was an increased accumulation of carbon on public reserved lands compared with the RPA Assessment scenario. At the national level the net effect of this change was a slightly higher rate of carbon accumulation in the 1990's and a moderation of the long-term decline in forest carbon accumulation rates calculated for the RPA scenario (Figure 4.8 and 4.12).

Private Lands

Despite a large reduction in public available timberland acreage, timber harvest removal from public forests is only 10 to 15 % less for any year of the projection than in the RPA scenario. Since removals from private lands far outweigh those from the public sector, the overall impact on private forest pools and flux is minimal relative to the RPA Assessment scenario previously described (Figures 4.8, 4.12).

The net accumulation of carbon in the forests for the period 1990-1999 is an increase of 44 Tg-C yr^{-1}, within 5% of the RPA scenario value of 46 Tg-C yr^{-1} for this period (Figure 4.12). After 2010, the 10-year mean changes in storage are negative, i.e., a net loss of carbon occurs. The rates of carbon loss are essentially the same for the two scenarios. At the end of the projection in year 2040, the total carbon storage of private forest lands is 22,393 Tg-C compared to 22,427 Tg-C for the RPA scenario, a difference of less than 1%.

Public Lands

Compared with the RPA Assessment scenario, current National Forest plans call for a substantial reduction in the acreage subject to timber harvest rotation (from 19% reserved in the RPA scenario to 41% reserved in the Current Forest Plans scenario). In the most extreme case, the Pacific Northwest West, this area reduction is approximately 57%; i.e., the total government-owned acreage in timber production falls from 51,194 ha in the RPA scenario to 37,289 ha. Forest acreage classified as reserved, i.e., deemed "unsuitable" for management harvest, more than doubles.

The reduction of acreage open to tree harvesting increases the net storage of carbon on public lands, particularly in the period 1990-2010. For example, both the RPA and Current Forest Plans scenarios express a 9 Tg-C yr^{-1} net accumulation of carbon from 1990 to 2000 for lands subject to harvest. The remainder of the accumulation is due to maturation of the reserved forest lands. With more than twice as much land in this category, the Current Forest Plans scenario has a total net carbon accumulation of 30 Tg-C yr^{-1} (Figure 4.12) during the 1990's, versus the 19 Tg-C yr^{-1} for the RPA scenario (Figure 4.8).

The difference in net accumulation between the scenarios is reduced over the course of the projection as the reserved forest stands mature and generally move into slower growing age classes. In year 2040, there is a 43 Tg-C yr^{-1} net accumulation associated with public forests, compared with a 32 Tg-C yr^{-1} accumulation for the RPA scenario. At this same time, total carbon

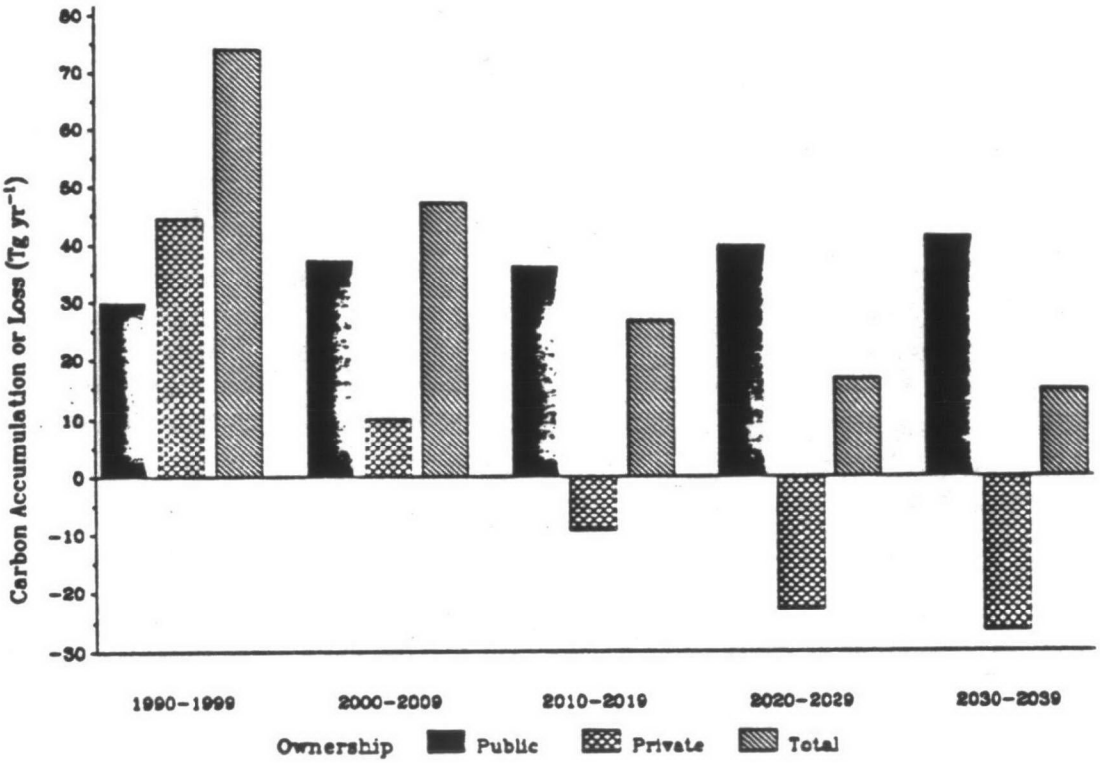

Figure 4.12. Projections of net carbon accumulation or loss (Tg-C yr^{-1}) for US timberlands (sum of available and reserved) for the Current Forest Plans scenario. Positive values are carbon accumulations. Negative values are carbon losses.

storage is estimated at 14,980 Tg-C versus 14,310 Tg-C, for the RPA scenario.

National Trends

The larger acreage of National Forests set aside from timber product management increases the overall net accumulation values associated with this scenario. For the period 1990-1999, the net accumulation of carbon is estimated at 75 Tg-C yr^{-1} (Figure 4.12), about 15% greater than for the RPA scenario. This accumulation declines over the projection period to 17 Tg-C yr^{-1} accumulation in 2040. The accumulation is always 9 to 14 Tg-C yr^{-1} greater than the RPA scenario (Figure 4.8).

Since the harvest levels for the nation as a whole do not differ greatly between the RPA Assessment and Current Plans scenarios (Figure 4.11, 4.13) there is unlikely to be a large difference in the pool of forest products still in use and in landfills. The analysis in Section 4.C.8 reports a difference between these scenarios of 130 Tg-C in this pool in the year 2040. The average difference in the sink strength over 50 years would thus be about 2.6 Tg-C yr^{-1}.

3. Reduced National Forest harvest scenario

In this preliminary analysis, there was little effect of the additional harvest reduction (0.36 billion cubic feet per year reduction in 2000) on National Forests. The growth and standing inventory for the available timberland in national forests over the 50-year period was nearly the same for the Current Forest Plans and Reduced National Forest Harvest scenarios. The bulk of the additional harvest reduction occurs in the Rocky Mountain Region, yet the projected inventory there was virtually unchanged. As with the Current Forest Plans scenario, there was not a sustained compensatory harvest increase on private lands.

A harvest reduction on public lands might be expected to result in higher standing biomass retained on that land base as indicted in the analysis of Harmon et al. (1990). The harvest reduction itself represents about 2.4 Tg-C yr^{-1} not removed from the public forests and, as described below, release of carbon via decomposition of stumps and woody debris is also avoided.

Because the estimates of future growth and growing stock volume were based on reports from individual National Forests, the detailed analysis of age class distributions needed to explain the insensitivity of the growth and growing stock inventory to harvest reductions was unavailable. Any firm conclusion regarding effects of these additional harvest reductions must therefore await more detailed treatment of long term shifts in stand age class distribution.

The case study analysis in the Pacific Northwest allowed an examination of the various factors which contribute to the net effect of harvest reductions. The initial net effect of reduced harvesting of timber in the Pacific Northwest was an increased accumulation in the forest of 1.1 Tg-C yr^{-1} per year. By year 2045 this effect had risen to 3.3 Tg-C yr^{-1} (Figure 4.14). The source from product decay dominated early in the scenario, but its contribution dropped to half of the total within 10 years. The harvested areas remained a source of carbon until year 2033 when the carbon sink from the area harvested early in the scenario began to exceed the new carbon source being created by harvesting.

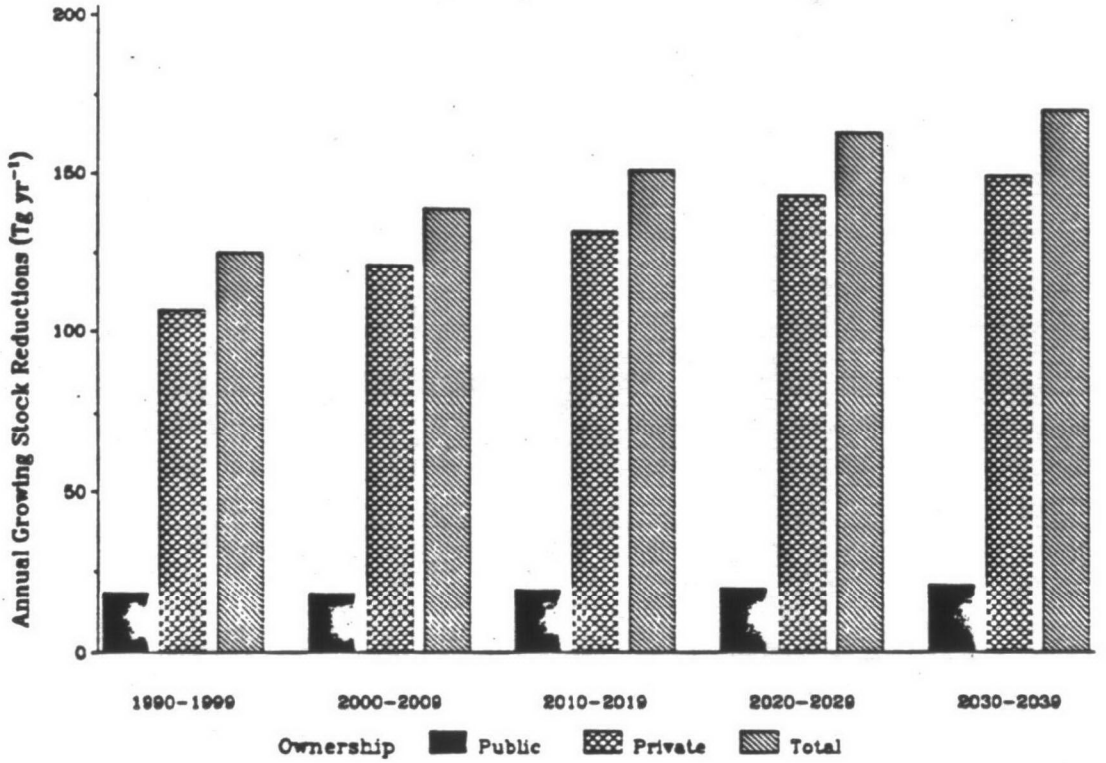

Figure 4.13. Projected annual growing stock volume reductions (Tg-C yr^{-1}) from US timberlands under Current Plans scenario.

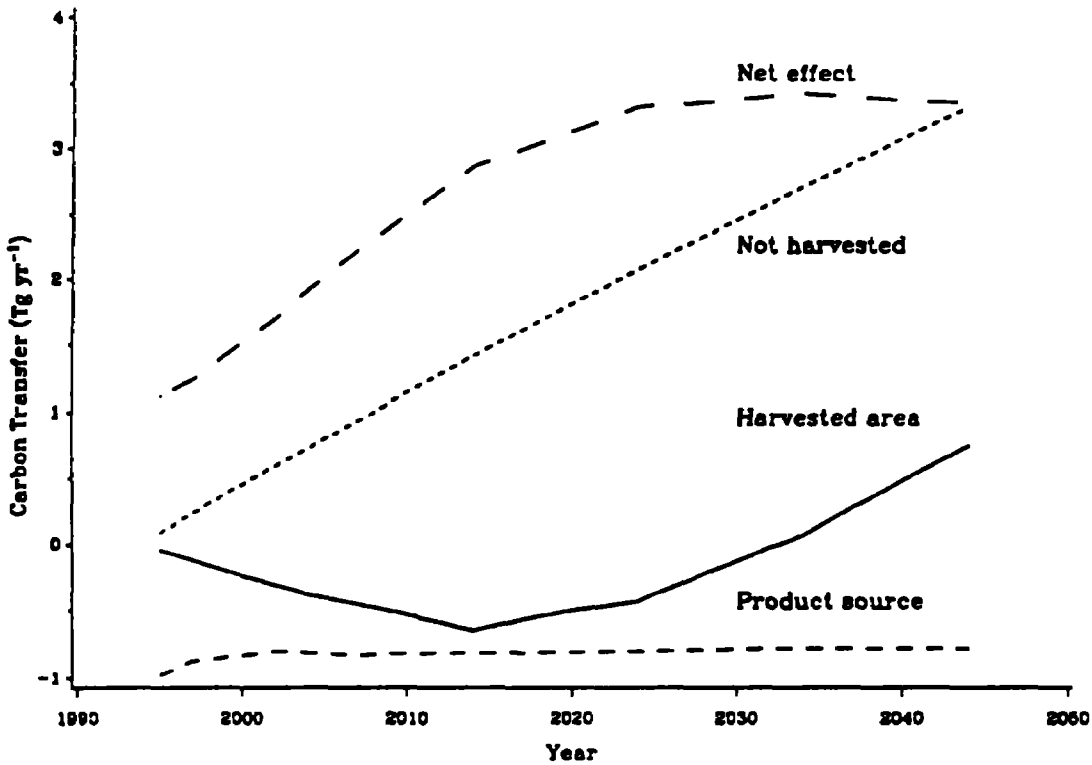

Figure 4.14. Change in carbon transfer (Tg-C yr^{-1}) to atmosphere over time from areas not harvested in the Pacific Northwest West Region. Values are based on differences between harvest in this region under the RPA scenario and the Reduced National Forest Harvest (0.293 x 10^9 cubic feet difference in 2000). Negative values are carbon sources created by harvesting. Positive values are carbon sinks maintained by not harvesting. The net effect is the total benefit of not harvesting in terms of carbon not emitted to the atmosphere.

The carbon sink which was eliminated by harvesting increased over time and became the largest factor in the net effect at year 2045.

This case study of the Pacific Northwest was intended to provide a framework for assessing the effects of harvest reduction on carbon flux between the atmosphere and land surface. The results suggest that savings in carbon emissions increase over time from harvest reduction and that over a 50-year period they remain relatively high.

The effects of harvest reductions in other regions would be different than in the Pacific Northwest because of differences in the harvest volume per unit area and in the time course of the total carbon increment for different forest types. Comparable studies in other regions would be of interest in evaluating a national policy with regard to harvest reduction on public lands.

4. Increased Recycling scenarios

The two recycling scenarios considered in these analyses assume a public lands management policy identical to that of the Current Forest Plans scenario; differences amongst these three scenarios are derived solely from the state of private timberlands. The High Recycling scenario assumes a rapid increase in the rate of wastepaper recycling between 1986 and 2000, and this is reflected in a rate of carbon accumulation on private timberlands of 8 Tg-C yr^{-1} greater than the Current Plans scenario in the 1990's (Figure 4.15). This difference peaks in the 2000's at 17 Tg-C yr^{-1}, but then declines nearly linearly to only 1 Tg-C yr^{-1} in the 2030's. The absolute carbon accumulation rate decreases continuously over the course of the simulation, from 82 Tg-C yr^{-1} in the 1990's to 15 Tg-C yr^{-1} in the 2030's.

The Moderate Recycling scenario models a more conservative increase in the recycling rate. In the 1990's its carbon accumulation rate is equivalent to the Current Forest Plans scenario. As the recycling rate increases over the next three decades, the carbon accumulation rate also rises relative to Current Forest Plans (Figure 4.15). By the 2020's the accumulation rate exceeds that of the Current Forest Plans by 13 Tg-C yr^{-1}, and also surpasses the accumulation rate of the High Recycling scenario in this and the succeeding decade. Here, too, the absolute accumulation rate declines with each decade from 74 Tg-C yr^{-1} in the 1990's to 25 Tg-C yr^{-1} in the 2030's.

The carbon sink benefits derived from the recycling scenarios are directly related to the reductions in harvest rates on the private timberland resulting from reuse of paper fibers. Although the rate of recycling assumed in the High Recycling scenario is always greater than that of the Moderate Recycling scenario (45% versus 28.5 to 41.5%), economic factors applied by the TAMM model, particularly projected stumpage prices and levels of lumber imports from Canada, dictate higher levels of timber harvest for the High Recycling scenario in the 2030's and 2040's. This determines the reversal in relative sink strength of these scenarios displayed in Figure 4.15. Integrated over the 50-year simulation, these accumulation rates produce 2040 forest carbon pools 0.3 Pg and 0.4 Pg higher than the Current Forest Plans scenario for the Moderate and High Recycling scenarios, respectively (Figure 4.16).

5. Afforestation scenarios

As in the case of the recycling

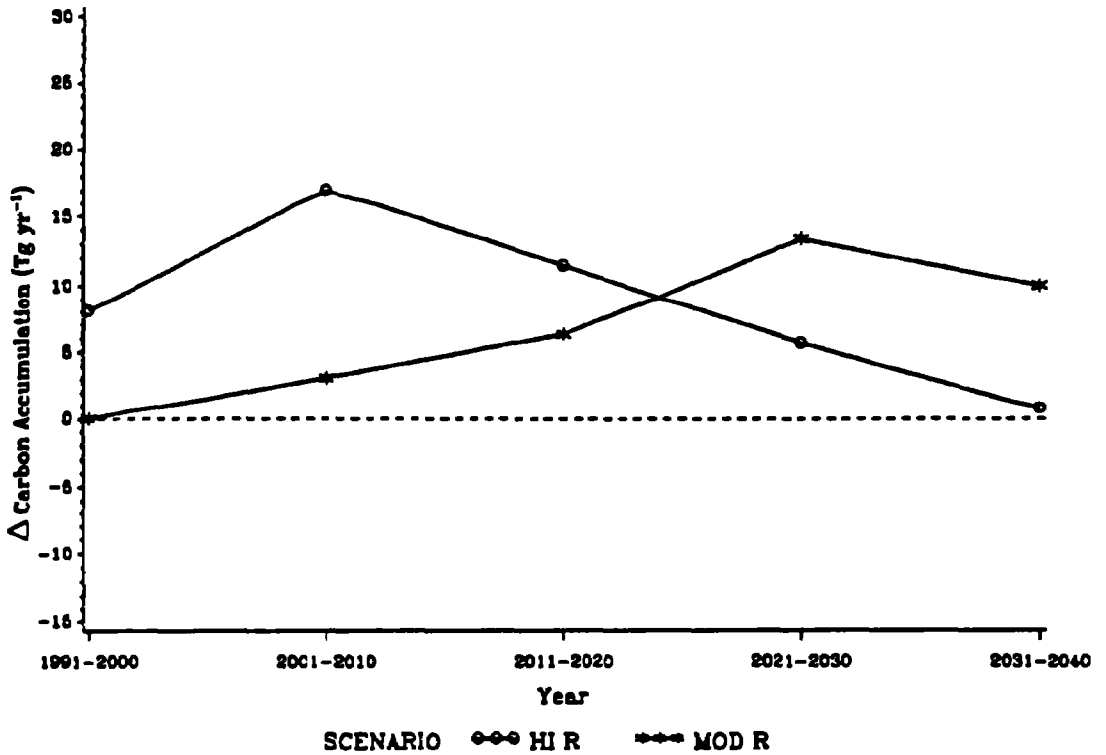

Figure 4.15. Annual carbon accumulation (Tg-C yr^{-1}) for recycling scenarios relative to Current Forest Plans projections. Scenario abbreviations are as follows: HI R = high recycling, MOD R = moderate recycling.

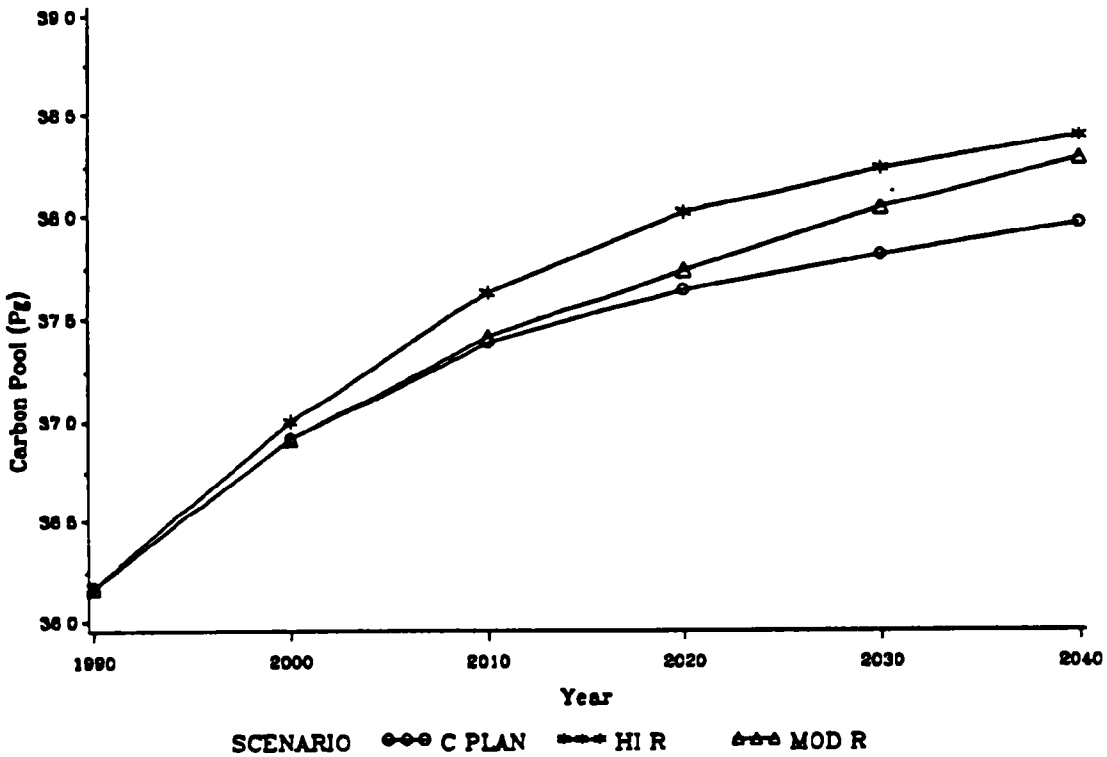

Figure 4.16. Comparison of projected total forest carbon pool (Pg-C) for the High and Moderate Recycling and Current Forest Plan scenarios. Scenario abbreviations are the same as in Figure 4.15. Scenario abbreviations are as follows: C Plan = Current Forest Plans, HI R = High Recycling, MOD R = Moderate Recycling.

scenarios public lands management under the afforestation scenarios conforms with the Current Forest Plans scenario, and differences arise only due to operations on private timberlands. Carbon accumulation is enhanced relative to the Current Plans scenario, reaching peak differences of 13 to 26 Tg-C yr^{-1} in the 2010's (Figure 4.17). The trend of the accumulation difference parallels growth rates of South Central loblolly pine plantations; the latter dominate the afforestation acreage allotments of all four scenarios. The accumulation rate in excess of the Current Forest Plans scenario declines over 50% over the next two decades, to 5 to 12 Tg-C yr^{-1}. As for all scenarios, absolute accumulation rates decline continuously over the course of the simulation, from 75 to 77 Tg-C yr^{-1} in the 1990's to 19 to 27 Tg-C yr^{-1} in the 2030's.

At the $110 million per year (for 10 years) funding level, the impacts of the Moulton and Richards and the Parks and Hardie scenarios are indistinguishable. At $220 million per year funding, the additional carbon accumulation rates achieved by the Parks and Hardie scenario are nearly double those of the Moulton and Richards approach (Figure 4.17). This result may reflect both the larger area planted (Table 3.8) and the regional distribution of the afforestation effort. The afforestation acreage in the South Central region is nearly equal in both scenarios. All other afforestation acreage in the Moulton and Richards scenario is located in the Rocky Mountains region. Additional acreage of the Parks and Hardie scenario is more widely distributed but occurs primarily in the North Central region, which would be expected to exhibit relatively higher growth rates.

Carbon pools at the end of the simulation (2040) are the highest predicted, ranging from 0.4 Pg ($110 million per year Moulton and Richards) to 0.8 Pg ($220 million per year Parks and Hardie) in excess of the Current Forest Plans scenario (Figure 4.18).

6. Combination scenarios

Unlike the recycling and afforestation scenarios, the combination scenarios assume public lands management identical to the Reduced National Forest Harvest scenario. The impact of the latter, in terms of carbon pools and accumulation rates, was nearly equivalent to the Current Forest Plans scenario. Therefore, the results of the first combination, joining Moderate Recycling with the Reduced National Forest Harvest, are very similar to the Moderate Recycling scenario previously described. The second combination, which adds the assumption of doubling of export levels, reduces the carbon sink effectiveness of moderate recycling, relative to the Current Forest Plans scenario, by 50% for decades in the period 2010 - 2040 (Figure 4.19 and 4.20).

Adding a $110 million per year (total $1.1 billion) afforestation effort more than tripled the carbon accumulation difference exhibited by the second combination relative to the Current Forest Plans. Here the increased paper recycling rate appears to more than compensate for increased exports. The absolute carbon accumulation rate is 42% above the afforestation scenario in the absence of these factors.

7. Scenario comparisons

The outcomes of these scenarios predict that US forests will accumulate carbon during each decade of the 50-year

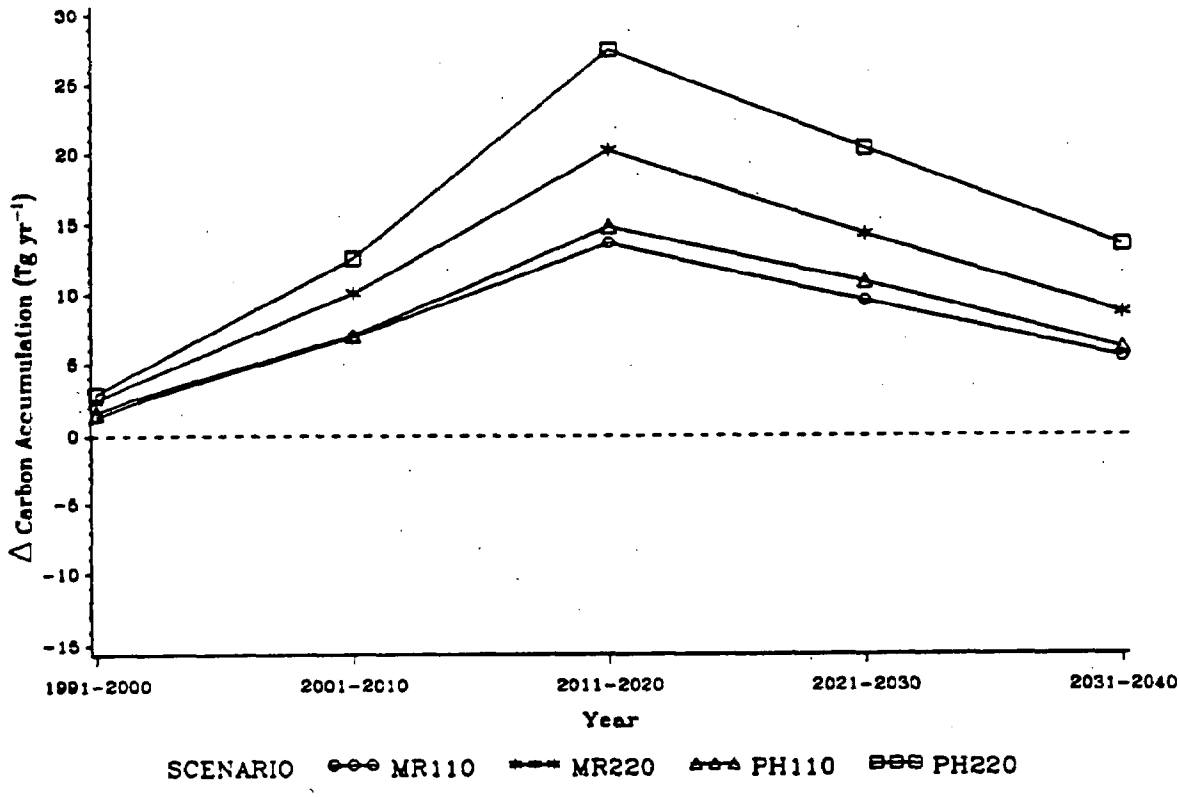

Figure 4.17. Annual carbon accumulation difference (Tg-C yr^{-1}) for afforestation scenarios relative to Current Fo11rest Plans projections. Scenario abbreviations are as follows: MR110 = Moulton and Richards, $110 million; MR220 = Moulton and Richards, $220 million; PH110 = Parks and Hardie, $110 million; PH220 = Parks and Hardie, $220 million.

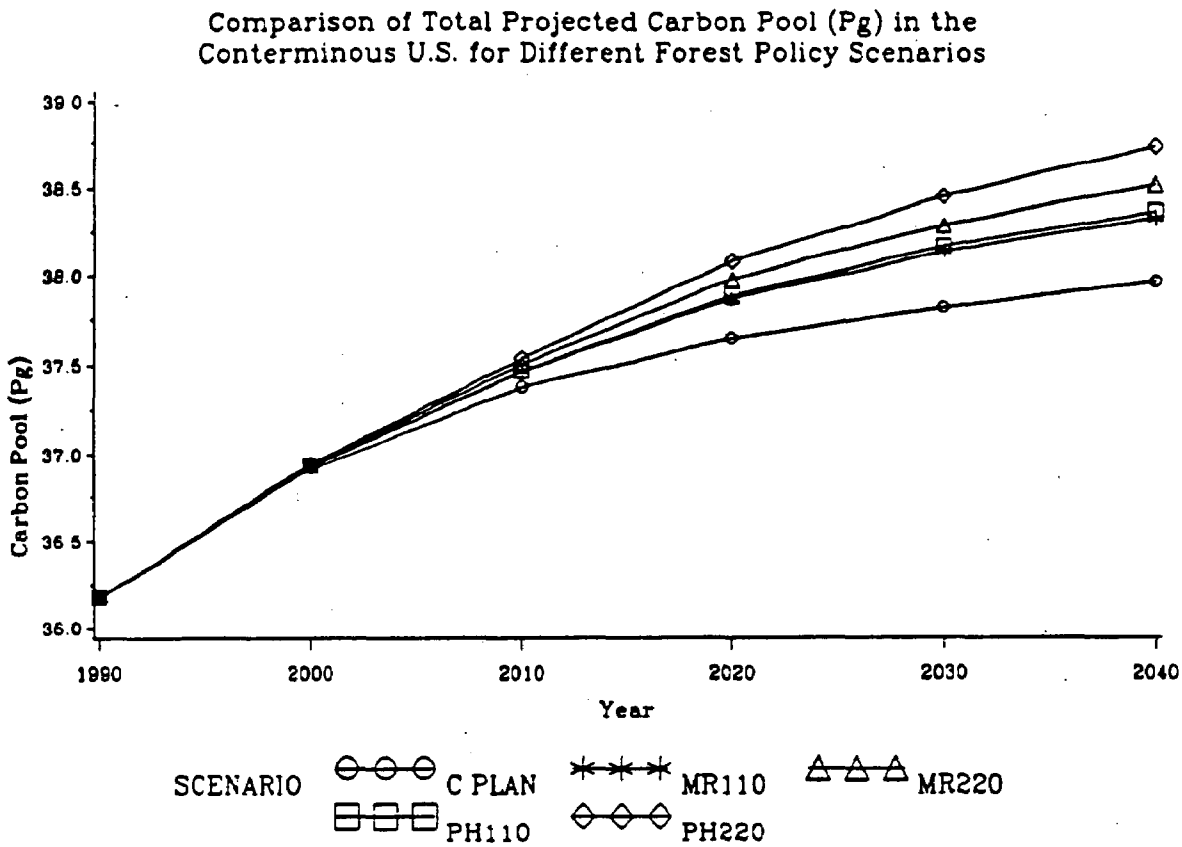

Figure 4.18. Comparison of projected total forest carbon pool (Pg-C) for afforestation scenarios. C Plan = Current Forest Plans scenario and abbreviations for the afforestation scenarios are the same as in Figures 4.16 and 4.17.

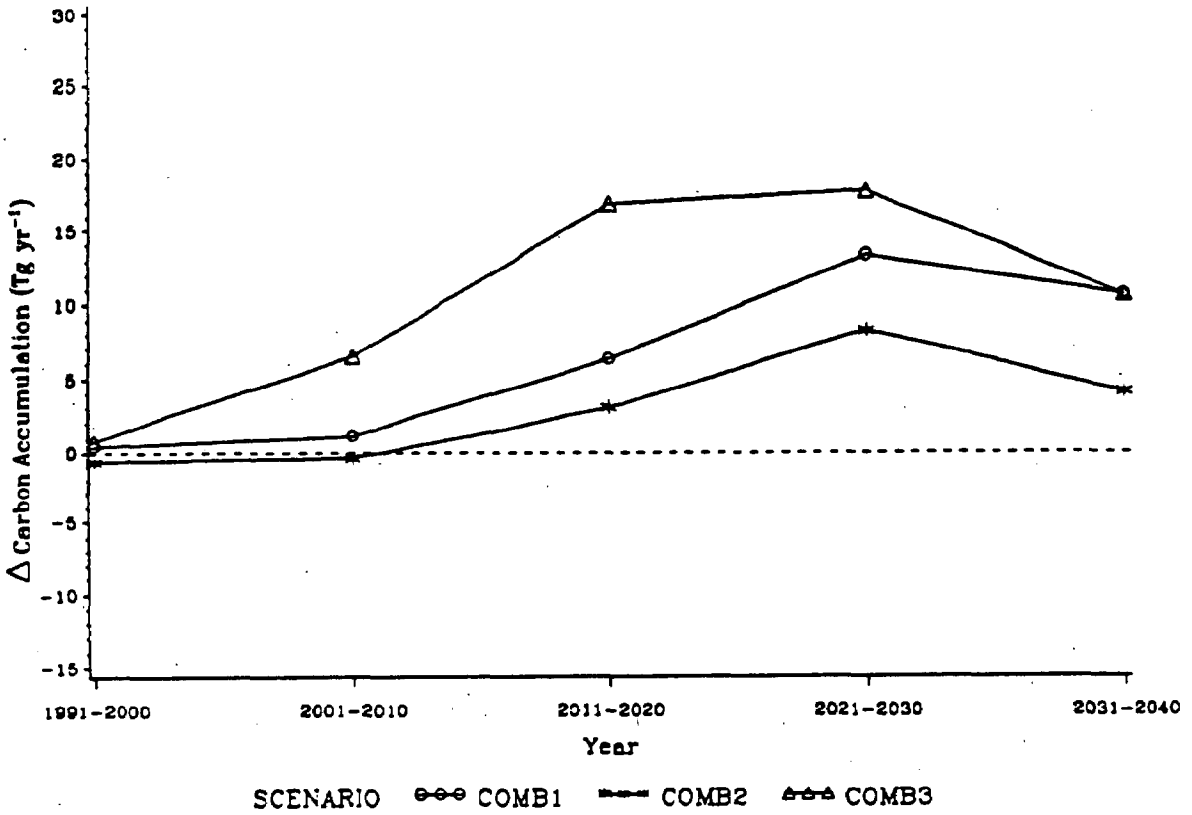

Figure 4.19. Annual carbon accumulation difference (Tg-C yr^{-1}) for combination scenarios relative to Current Forest Plans projections. Scenario abbreviations are as follows: COMB1 = Combination 1, COMB2 = Combination 2, and COMB3 = Combination 3.

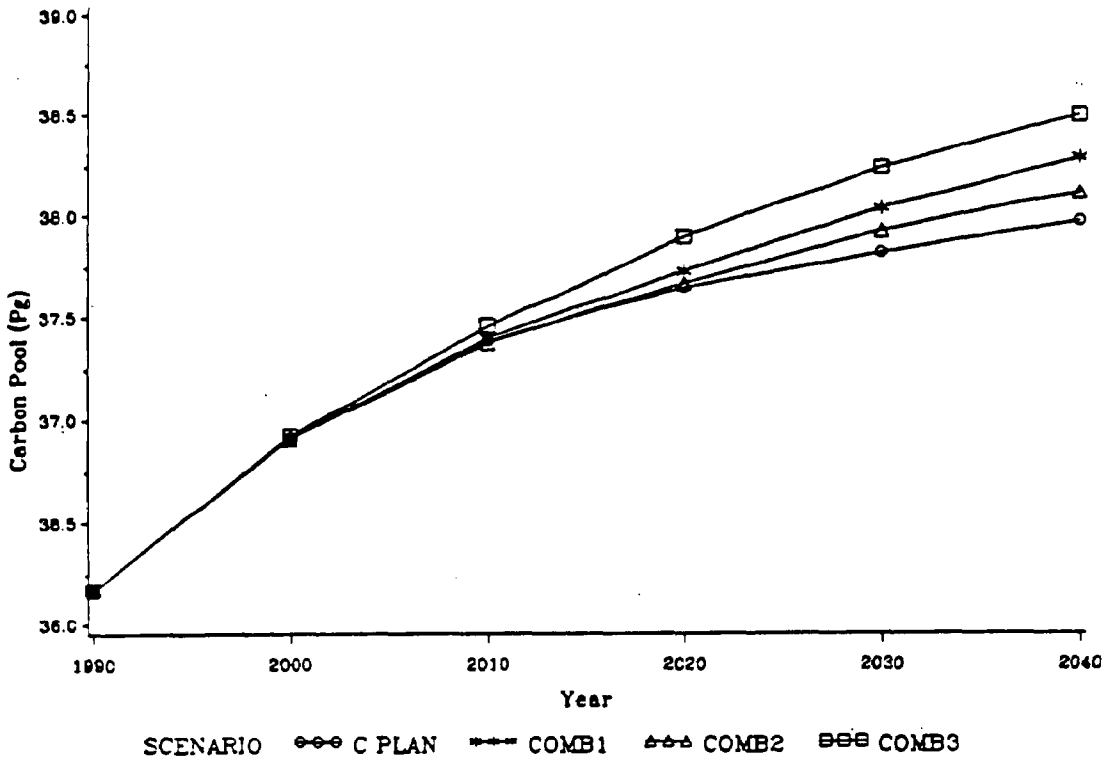

Figure 4.20. Comparison of projected total forest carbon pool (Pg-C) for the combination scenarios and the Current Forest Plans scenario. Scenario abbreviations are the same as in Figures 4.18 and 4.19.

period (Table 4.2). The rate of carbon accumulation will, however, decline dramatically over this same period. Each scenario predicts that the private forests, as simulated by the TAMM economics model coupled with the ATLAS timber inventory projection model, will change from accumulating carbon to losing carbon between 2010 and 2040 (Figure 4.12). As noted earlier, contributing factors include rising harvest rates, shorter average rotation lengths, and the conversion of the land base to other uses such as urbanization. Less rigorous data and models were available to project public forest inventories. Both available and reserved public timberlands appear to act as carbon sinks throughout the simulation period and more than balance carbon losses from private timberlands in later decades (Figure 4.12).

Although these trends are universal among the scenarios, there are important distinctions which contribute to differences in carbon pool totals and in temporal variations in carbon accumulation or loss. The 1990 total carbon pool of US forests is estimated to be 36.2 Pg-C. Under the now obsolete RPA Scenario, this figure is predicted to rise to 37.6 Pg-C in 2040 (Figure 4.16). Increased carbon storage in public forests accounts for all of this difference. Private timberlands have increased total carbon at the midpoint of the simulation, but the total proceeds to decline to the 1990 level by the simulation's end.

The Current Forest Plans mark a significant departure from the RPA Scenario. A substantial portion of the public forest land base is shifted from the available category, i.e. forests subject to harvest, into long-term preserves. As a result, carbon accumulation in public forests is increased, with rates 7 to 14 Tg-C yr^{-1} higher than the RPA Scenario over the course of the simulation.

Harvest rates on private forests increased only slightly in response to the change in public forest management policies. From 1990-2010, carbon accumulation rate on private lands is only 2 Tg-C yr^{-1} less than the RPA Scenario. However, this increased rate of harvest is not sustained, and by the 2030's the carbon accumulation rate is 1 Tg-C yr^{-1} in excess of the RPA scenario. For 2040, the total US forest carbon pool is estimated at 38.0 Pg. The carbon total for private timberlands changes less than 0.1 Pg during the period 1990-2040.

Both the recycling scenarios and the afforestation scenarios further enhance carbon accumulation over the 50-year simulation period. The afforestation scenarios have greater impact on the 2040 carbon pool total: values range from 38.3 to 38.7 Pg-C versus 38.3 to 38.4 for the recycling policies (Figures 4.16, 4.18). However, the High Recycling scenario manifests the largest immediate increase in carbon accumulation. For the period 1990 to 2010, carbon accumulation rates are an average of 13 Tg-C yr^{-1} higher than for the Current Forest Plans (Figure 4.15). By comparison, the most effective afforestation policy over this same period, the $220 million per year Parks and Hardie scenario, achieves a 8 Tg-C yr^{-1} increase (Figure 4.17). The major benefits of afforestation policies are delayed until the latter half of the simulation as the growth rates of the young forests begin to substantially increase. For the period 2010 to 2040, the average accumulation rate of the $220 million Parks and Hardie scenario is 19 Tg-C yr^{-1} greater than the Current Forest Plans accumulation rate (Figure 4.17). The High Recycling scenario

Table 4.2. Summary of policy scenario results.

1989 RPA Assessment	Forest land accumulates carbon at a rate of 66 Mt-C per year in the 1990's after harvest removals are accounted for. The accumulation rate declines over time to 5 Mt-C per year because of a shrinking private forest land base, increased harvest levels and a shift to younger age classes.
Current Forest Plans (base scenario)	Public lands accumulate an additional 10 Mt-C per year, bringing the national sink to 74 Mt-C per year in the 1990's. The downward trend over time in the carbon accumulation rate is similar to the RPA Assessment Scenario.
Reduced National Forest Harvest	Harvest reductions are likely to retain an additional several Mt-C per year on public lands.
Moderate Recycling	Carbon uptake, relative to Current Forest Plans, gradually increases to a peak of 13 Mt-C per year greater in 2020's. Carbon pool increase by 2040 is 75% of that achieved by High Recycling.
High Recycling	Reduced harvests on private lands lead to a carbon accumulation rate of 8 Mt-C per year greater than the Current Forest Plans in the 1990's, and 17 Mt per year greater in the 2000's. The difference declines in the later decades, and the same trend of reduction in the carbon accumulation rate over time is observed. Carbon pool in 2040 is 0.4 Gt greater than Current Forest Plans estimate.
Afforestation Moulton and Richards $110 million	Afforestation in 1990's enhances carbon accumulation relative to Current Forest Plans by 13 Mt-C per year in 2010's. Carbon pool value in 2040 is 0.4 Gt greater.
Afforestation Moulton and Richards $220 million	Carbon accumulation gains over Current Forest Plans are about 150% of the gain from the $110 million per year scenario.
Afforestation Parks and Hardie $110 million	Essentially the same result as the Moulton and Richards $110 million scenario.
Afforestation Parks and Hardie $220 million	Most effective of the afforestation scenarios. Carbon pool in 2040 is 0.8 Gt greater than Current Forest Plans.
Combination 1	Very similar to Moderate Recycling.
Combination 2	Reduces the carbon accumulation rates of Combination 1, in excess of Current Forest Plans, by 50% from 2010 to 2040.
Combination 3	Increases the carbon accumulation rate from Combination 2. Carbon pool in the year 2040 is 0.5 Gt greater than the Current Forest Plans scenario.

achieves only a 6 Tg-C yr^{-1} advantage over the Current Forest Plans scenario during this same period (Figure 4.15).

The combination scenarios add relatively little information. As has been mentioned, the first combination is essentially the same as the Moderate Recycling scenario, in part due to the apparently negligible effect of lower public forest harvest rates on the forest inventory (combination scenarios assume public lands management described as the Reduced National Forest Harvest). The third combination, combining the Moderate Recycling assumption with doubled export levels and the $110 million per year Moulton and Richards Scenario, yielded a benefit significantly greater than the unaltered afforestation scenario. The increased timber supply associated with paper recycling and afforestation was greater than the harvest increases associated with the doubling of timber exports. The combination scenarios emphasized the significant impact otherwise unconsidered factors, in this case timber export trends, may have upon the simulations described in this report.

8. Results of HARVCARB analysis

In the Current Forest Plans scenario, HARVCARB allocated 81% of each year's harvest during the first decade to storage in products or landfills (Table 4.3). The balance of harvested carbon returns to the atmosphere immediately, primarily from use for energy. To the extent that it substitutes for energy from fossil fuel use, the use of forest-products for energy can be considered to be a permanent carbon sink or an energy offset.

In succeeding years, carbon from current harvests continues to be allocated largely to products. However, there is also movement of carbon from the products pool to both landfills and the atmosphere as older products are retired. By the year 2040, 64% of the carbon harvested to that point is still in products or landfills.

Because of a lack of inventory data on the amount of forest carbon currently in products and landfills, the absolute magnitude of the current and future products pool cannot be known with much certainty. However, the relative effect of different forest policy scenarios initiated at the current time can be evaluated using HARVCARB. Tables 4.4 and 4.5 indicate the differences in the forest-products, landfill, and energy-offset pools under two of the forest policy scenarios compared to the Current Forest Plans scenario. Unlike all other scenarios, the RPA scenario results in more carbon stored in products and landfills because projected harvests are higher.

The effect of the alternative scenarios, compared to the Current Forest Plans scenario, can be summarized as the 50-year average annual change in effective carbon storage (Table 4.6). This value was calculated as the sum of the difference in the product, landfill, and energy-offset pools in 2040, minus the difference in the non-energy related release to the atmosphere, divided by 50. For example, the High Recycling scenario forecasts a change in the product, landfill, and energy-offset pools for 2040 of -58.6, -44.7 and -26.8 Tg, respectively. Thus, the total change in these pools is -130 Tg. This is partially offset by a change of 33.9 Tg-C in the amount of carbon released to the atmosphere and not used for energy. Thus, the net average change in effective carbon storage is (-130 - (-33.9))/ 50, or -1.9 Tg-C yr^{-1}.

Table 4.3. Cumulative disposition of carbon (Tg-C) for the Current Forest Plans scenario.

Year	Products still in-use	Still in landfills and dumps	To atmosphere: Used for energy	To atmosphere: Not used for energy
1980	0.0	0.0	0.0	0.0
1985	499.4	168.0	101.2	4.3
1990	959.3	375.7	255.1	60.6
1995	1360.9	588.3	420.3	205.7
2000	1764.9	821.8	598.8	381.5
2005	2174.0	1078.4	789.2	573.8
2010	2585.5	1360.9	990.1	781.1
2015	2988.4	1673.7	1205.3	1002.4
2020	3380.0	2011.8	1433.8	1236.3
2025	3770.7	2373.7	1672.9	1483.1
2030	4150.7	2756.0	1920.2	1741.8
2035	4516.8	3157.8	2173.7	2011.2
2040	4861.7	3577.6	2431.9	2288.4

Table 4.4. Difference in carbon disposition (Tg-C) between the RPA scenario and the Current Forest Plans scenario. Positive numbers indicate more allocation of carbon than for the Current Forest Plans scenario.

Year	Products still in-use	Still in landfills and dumps	To atmosphere: Used for energy	To atmosphere: Not used for energy
1980	0.0	0.0	0.0	0.0
1985	0.2	0.1	0.0	0.0
1990	2.3	0.7	0.5	0.0
1995	4.9	1.9	1.2	0.1
2000	11.7	5.4	3.9	0.5
2005	18.2	9.1	7.0	2.8
2010	27.8	14.6	11.3	5.8
2015	39.6	21.3	16.5	9.7
2020	47.0	27.3	21.1	14.8
2025	53.3	33.3	25.5	20.3
2030	59.3	39.7	30.4	25.6
2035	68.6	47.7	36.0	30.7
2040	75.0	55.5	41.4	36.3

Table 4.5. Difference in carbon disposition (Tg-C) between the High Recycling scenario and the Current Forest Plans scenario. Negative numbers indicate less allocation of carbon than for the Current Forest Plans scenario.

Year	Products still in-use	Still in landfills and dumps	To atmosphere: Used for energy	To atmosphere: Not used for energy
1980	0.0	0.0	0.0	0.0
1985	-0.4	0.0	0.0	0.0
1990	-2.8	-0.9	-0.7	0.0
1995	-3.1	-0.5	+0.2	-0.3
2000	-21.9	-5.7	-1.9	-0.6
2005	-49.8	-15.7	-7.9	-2.3
2010	-67.3	-24.2	-14.0	-7.9
2015	-76.7	-31.0	-19.4	-15.4
2020	-81.6	- 36.5	-23.8	-22.0
2025	-82.6	-40.9	-26.9	-27.3
2030	-80.3	-44.4	-28.5	-30.9
2035	-71.0	-45.5	-28.4	-33.3
2040	-58.6	- 44.7	-26.8	-33.9

Table 4.6. The 50-year average change in flux to product, landfill, and energy-offset pools for the four alternative policy scenarios, compared to the Current Forest Plans scenario. A plus (+) sign means an additional carbon sink.

Policy Scenario	Average Change in Sink ($Tg\text{-}C\ yr^{-1}$): Total Carbon	Average Change in Sink ($Tg\text{-}C\ yr^{-1}$): CH_4 Corrected
RPA	+2.7	-0.2
Reduced National Forest Harvest	-1.2	+2.4
Moderate Recycling	-0.7	-0.5
High Recycling	-1.9	+0.8
Afforestation		
Moulton & Richards $110 million	+1.7	-0.2
Moulton & Richards $220 million	+2.4	0
Parks and Hardie $110 million	+1.7	-0.2
Parks and Hardie $220 million	+2.4	0
Combination 1	-1.1	-1.1
Combination 2	-0.9	+0.6
Combination 3	-1.6	+0.8

The changes in effective carbon storage generally followed changes in harvest levels. The Afforestation scenarios had somewhat higher harvests than the Current Forest Plans scenario and have increased carbon sequestered in products and landsfills.

From a strict carbon-accounting point-of-view, the decreases in storage in the product, landfill, and energy-offset pools associated with other scenarios partially cancel the increases in the carbon sink on the forest land base; i.e., to a certain extent, these scenarios shift carbon from the products, landfills, and energy-offset pools to the forests. It should be noted, however, that about half the carbon released from landfills is in the form of methane (EPA 1991), which, molecule for molecule, is about 25 times more effective than carbon dioxide as a greenhouse gas (NAS 1991). The difference in the global warming potential of each emitted molecule of a greenhouse gas also depends on its atmospheric lifetime. Integrated over 100 years the global warming potential of emitted methane is about 10 times greater than emitted CO_2 (IPCC 1990).

The effect of methane emissions can be put into perspective by giving half the non-energy emissions a weight of 10. For example, for the High Recycling scenario, the non-energy release would change from -33.9 Tg-C to -(16.9 * 10), or -169 Tg. Using this value, the net average change in flux becomes (-130 - (-169))/50, or 0.8 Tg-C yr^{-1}. In other words, from a climate change point-of-view, the High Recycling scenario creates an effective carbon sink of 0.8 Tg-C yr^{-1} by reducing methane emissions. This should be added to the forest carbon sink to estimate the total impact of this scenario compared to the Current Forest Plans scenario. Correcting for methane has similar effects on the average flux associated with the other scenarios (Table 4.6). When the decreased landfill methane emissions associated with the alternative scenarios are viewed in this way, the benefits in terms of reduced creation of warming potential are generally greater than the loss of a carbon sink in the forest products and landfill pools. These results emphasize the importance of accounting for the post-harvest disposition of forest products, both in terms of carbon and in terms of potential climate impact.

D. Discussion

The dramatic influence of human activity on the global carbon cycle has been well documented (IPCC 1990). As understanding increases concerning the mechanisms involved in anthropogenic perturbation of the carbon cycle, opportunities will develop to apply policies for managing these human influences. The basis for policy decisions must in part be quantification of potential benefits of particular policy options, and the present study is a step in that direction.

The analyses presented here suggest a current potential rate of carbon accumulation (considering only plant growth and decay) on timberlands in the US of 286 Tg-C yr^{-1}. This rate of potential accumulation is possible only because of the continuous removal of carbon by harvesting and the resetting of the stand development cycle. Under unmanaged conditions the rate of carbon accumulation on the land surface would be lower or even negative because fire and increased rates of woody debris decomposition would balance uptake. Nevertheless, the value of 286 Tg-C yr^{-1} indicates the current short-term carbon sink in the

absence of wood harvesting.

When the effects of harvesting and fire are considered, the net carbon accumulation on the forested land surface under current conditions is about 78 Tg-C yr^{-1}. The corresponding net carbon accumulation in forest products and landfills is at least 36 Tg-C yr^{-1}. By way of comparison, the forest sector in Canada is also thought to be a small carbon sink (25 Tg-C yr^{-1} on the forest land base and 21 Tg-C yr^{-1} in products). Interestingly, the tree biomass pool is decreasing and the carbon accumulation on the land base appears to be primarily in the pool of woody debris and soil carbon (Kurz et al., 1992).

Under the assumptions of the 1989 RPA Assessment, which called for maintaining a relatively high level of harvesting on the National Forests but showed a long-term reduction in the land base for private timberland, the net carbon accumulation rate for US forests decreases over time. There appears to be a close relationship between the increase in harvest (about 50 Tg-C yr^{-1} from 1990-2040) and the decrease in the accumulation rate (decrease of about 60 Tg-C yr^{-1} from 1990-2040). Thus, forest sector policies which tend to reduce harvests, such as increased paper recycling and direct reductions in National Forest harvest levels, are likely to increase the rate of carbon accumulation on the forest land base. Note that the carbon accumulation rate in forest products still in use and in landfills would decrease with harvest reduction. However, the magnitude of that decrease is small relative to the benefit achieved in terms of increased carbon accumulation on the forest land base. When methane emissions from landfills are accounted for, decreased accumulation in landfills is a net benefit with respect to potential global warming.

Changes in the forest land base represent the other major option for increased carbon sequestration covered in this analysis. In the scenarios considered here, the maximum increase in the forest land base was 5 million ha, and that level of afforestation resulted in carbon sink averaging nearly 15 Tg-C yr^{-1} over the 50-year simulation. Nationally, however, there are approximately 47 million ha of marginally productive, privately-owned crop and pasture land that is biologically capable of supporting tree growth (Parks 1992). Thus, the maximum afforestation scenario considered in this report represents only 10% of land potentially available. This land presents an important opportunity for expanding the area of forest land in the US and thereby increasing the rate of carbon sequestration.

Other strategies for increasing carbon storage by forests include: (1) clearing and regenerating poorly stocked forest land if current productivity is well below average; (2) applying intermediate stand treatments if the current stand is overstocked to the point of stagnation; (3) carefully evaluating harvest and regeneration of mature forests using the longest possible time frame; (4) and managing for longer rotation lengths (Birdsey 1992b). In the case of longer rotation lengths, an upper limit on living forest biomass is eventually reached and additional tree growth is largely offset by mortality. Thereafter, the net additions to the terrestrial carbon storage become marginal, except in those systems where woody debris accumulation continues.

As a more permanent approach to offsetting carbon dioxide buildup in the

atmosphere, woody biomass crops might be used as substitutes for fossil fuels in energy production. Carbon is cycled between the biosphere and the atmosphere in that scheme, rather than being emitted in a one way flux from fossil fuel combustion. Assuming that optimal conversion and production technologies are applied, and given a conservative estimate of the available US land base, biomass energy could displace as much as 6% of current annual carbon emissions (Wright et al. 1992). Technological advancements in the future could result in displacement of as much as 20% of fossil fuel carbon emissions (Wright et al. 1992). There would also be a short term sink during establishment of energy plantations.

Substitution of wood for materials such as cement, aluminum, and plastics which use considerable energy in their manufacture, or emit CO_2 as a byproduct of manufacture, is another approach to reducing net CO_2 emissions (CORRIM 1976). The degree to which such substitution is made depends in part on the cost of wood. Under the recycling and afforestation scenarios projected stumpage price in the South are driven down considerably. This trend would promote wood substitution and reduce CO_2 emissions. We have not attempted to evaluate the reciprocial situation, where reductions in harvest promote use of energy intensive wood subsitutes. Additional carbon sinks related to management of the biosphere include reconversion of farmland to wetland and modified agricultural practices to reduce the loss of soil carbon (Johnson and Kern 1991).

Considered as a whole, there are significant opportunities for conservation and sequestration of carbon in the forestry sector of the US and globally (Samuelson and Hair 1992). Obviously, forest sector policies alone will not compensate for the fossil carbon emission increases projected over the coming decades (Lashof and Tirpak 1990, IPPC 1992a). However, they can contribute to moderating the overall trend towards increasing CO_2 conentration.

E. Presentation of results for all scenarios

Under a business as usual scenario, US fossil emissions are expected to increase at a rate of about 6-10 Tg-C yr^{-1}. The resulting emission rate in the year 2000 would thus be nearly 1,400 Tg-C yr^{-1}. Assuming a goal of stabilizing US emissions at the 1990 level by the year 2000, fossil emissions would need to begin decreasing within the next few years and return to the 1,300 Tg-C yr^{-1} by the year 2000. As described in this study, the forest sector presents a variety of options by which to increase carbon sequestration in the US and thus offset fossil carbon emissions. The carbon benefits might be credited in a variety of ways, and the most appropriate format for results of the scenarios depends on how the results will be used.

The first format for presentation of the scenario comparisons is simply the total forest carbon pool over time, including vegetation, litter, woody debris, and soil carbon. Results from several of the scenarios are presented in this format in Figure 4.21. Care must be taken with this approach to avoid artificial variations in the soil carbon associated with changes in the forest land base. The data in Figure 4.21 allow only for soil carbon gains associated with afforestation.

A second format presents the time course for the change in the carbon pool over the 50-year simulation

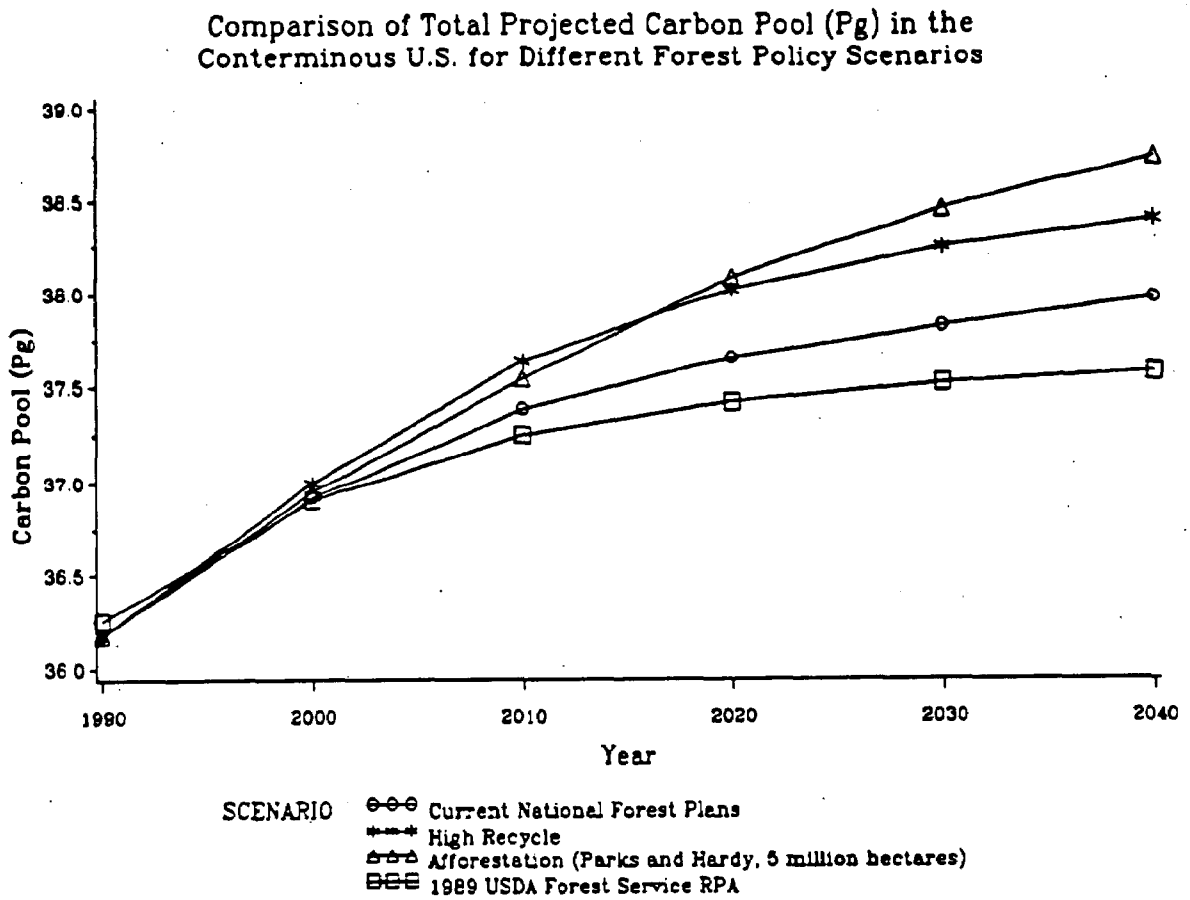

Figure 4.21. Scenario comparison based on projected total forest carbon pool (Pg-C). Abbreviations are as follows: C Plan = Current Forest Plans, HI R = High Recycling, PH220 = Afforestation (Parks and Hardie $220 Million), RPA = 1989 RPA Assessment.

Table 4.7. Undiscounted and discounted carbon benefit for eleven forest policy scenarios. Values are the increased carbon storage achieved over the 50 year simulation compared to the Current Forest Plans scenario. Units are million tons (Mt = Tg) of carbon. Abbreviations: MR110 and MR220 are Moulton and Richards $110 million and $220 million afforestation scenarios; PH110 and PH220 are Parks and Hardie afforestation scenarios.

		Discount Rate (%)			
Scenario	Undiscounted	1	2	5	10
RPA	-474	-454	-358	-261	-187
Reduced NF Harvest	-17	-15	-8	-2	1
Moderate Recycling	328	310	203	105	43
High Recycling	429	411	354	279	208
MR110	369	350	259	164	91
MR220	556	527	390	246	137
PH110	404	383	283	117	97
PH220	766	727	529	327	176
Combination 1	319	302	192	95	36
Combination 2	143	135	82	34	5
Combination 3	525	497	348	200	96

(Figure 4.22). The value at each time is the difference in the total carbon pool between a reference scenario and the specified policy scenario. The Current Forest Plans scenario is used as the reference scenario because it reflects the most up-to-date assumptions about harvests on federal lands. As an example of its utility, this format reveals the comparatively rapid accumulation of benefits from the High Recycling scenario compared to the afforestation scenarios.

A third approach compares the scenarios in terms of the total benefit over the course of the 50 year simulation. The carbon benefit could be applied to offset emissions in excess of the stabilization level either before or after the year 2000. To reflect the increasing uncertainty over time associated with model results, a sensitivity analysis has been made using annually compounded discount rates. For the present study, we have used discount rates beginning in the year 2000, of 10, 5, 2 and 1% (Table 4.7, Figure 4.23). There is an interaction of the timing of the benefits and the magnitude of the discount rate such that ranking of the scenarios may change in some cases.

A fourth presentation of the results is in terms of differences in the annual carbon sink over time (Figure 4.24). This format again indicates the benefits achieved by the policy scenarios without regard to the overall source or sink of the forest sector. This approach allows for ready comparison to rates of fossil emissions.

A fifth approach to presenting the

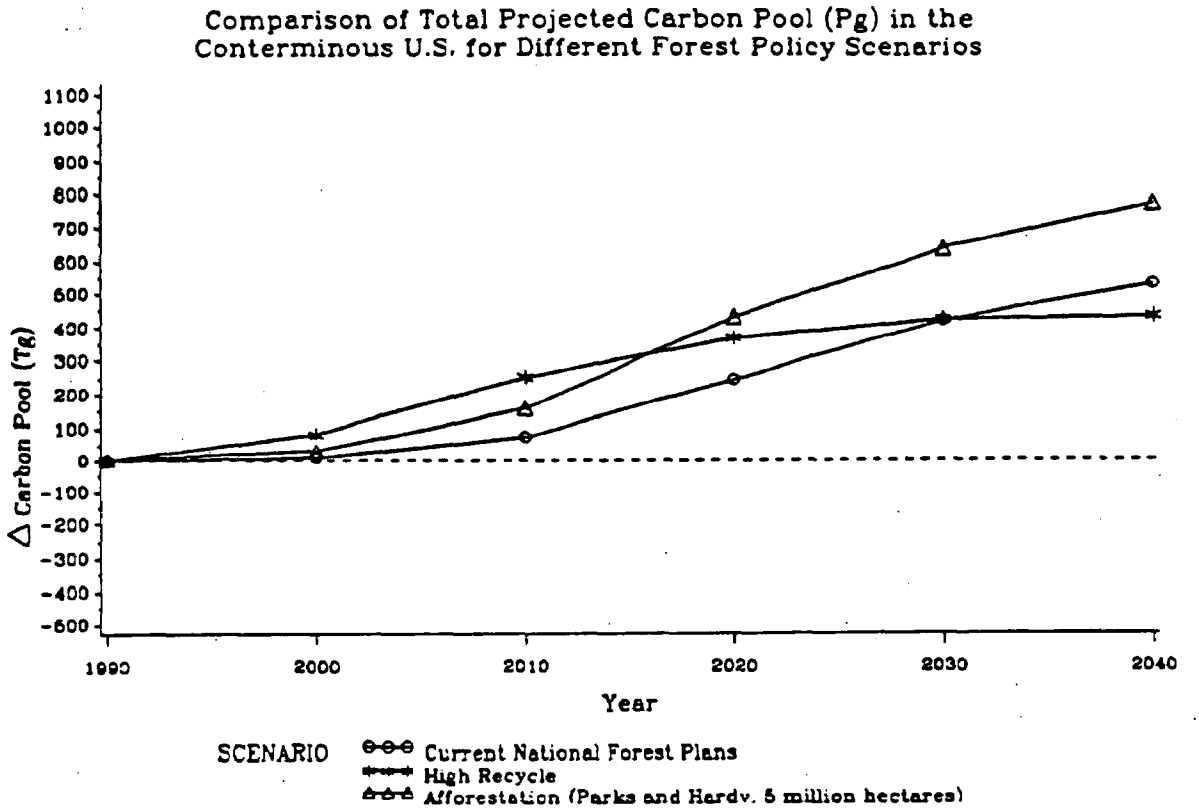

Figure 4.22. Scenario comparison based on difference from the Current Forest Plans scenario in projected total forest carbon pool (Tg-C). The abbreviations are as follows: HI R = High Recycling, PH220 = Afforestation (Parks and Hardie, $220 Million), COMB3 = Combination 3.

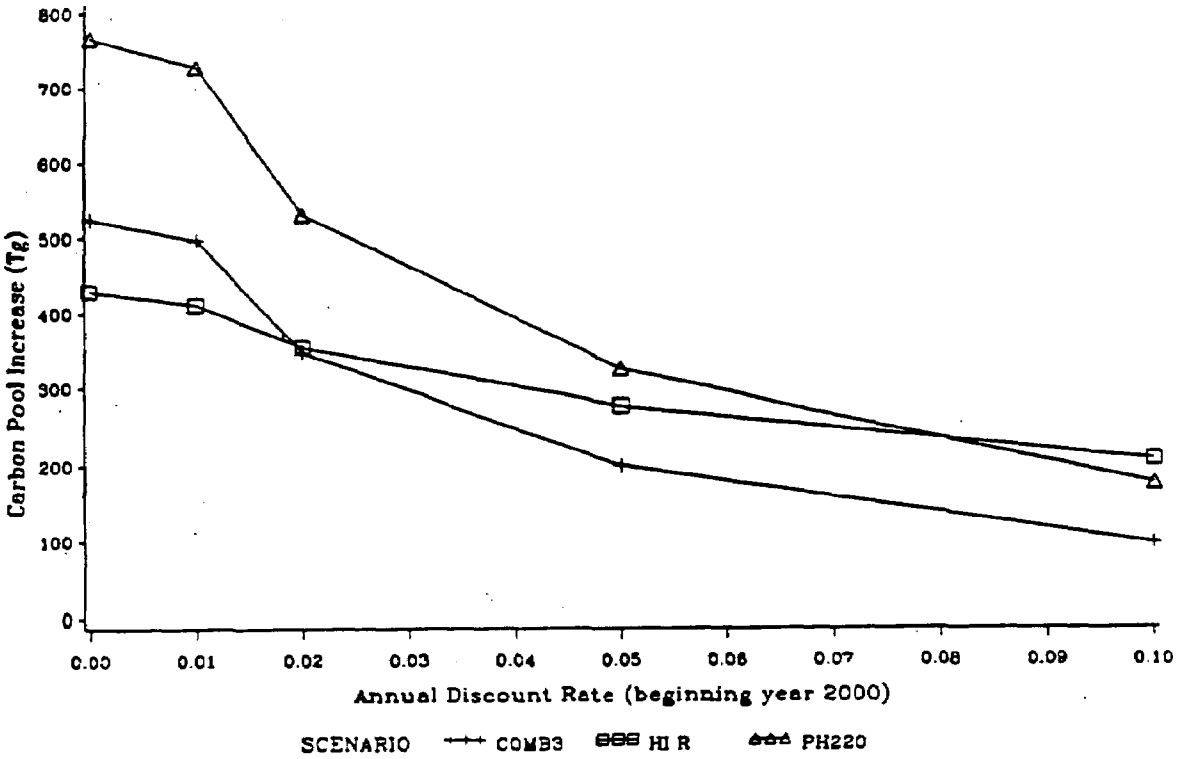

Figure 4.23. Effects of assumed discount rate on the carbon sequestration benefit (Tg-C over 50 years). Abbreviations are the same as in Figure 4.22.

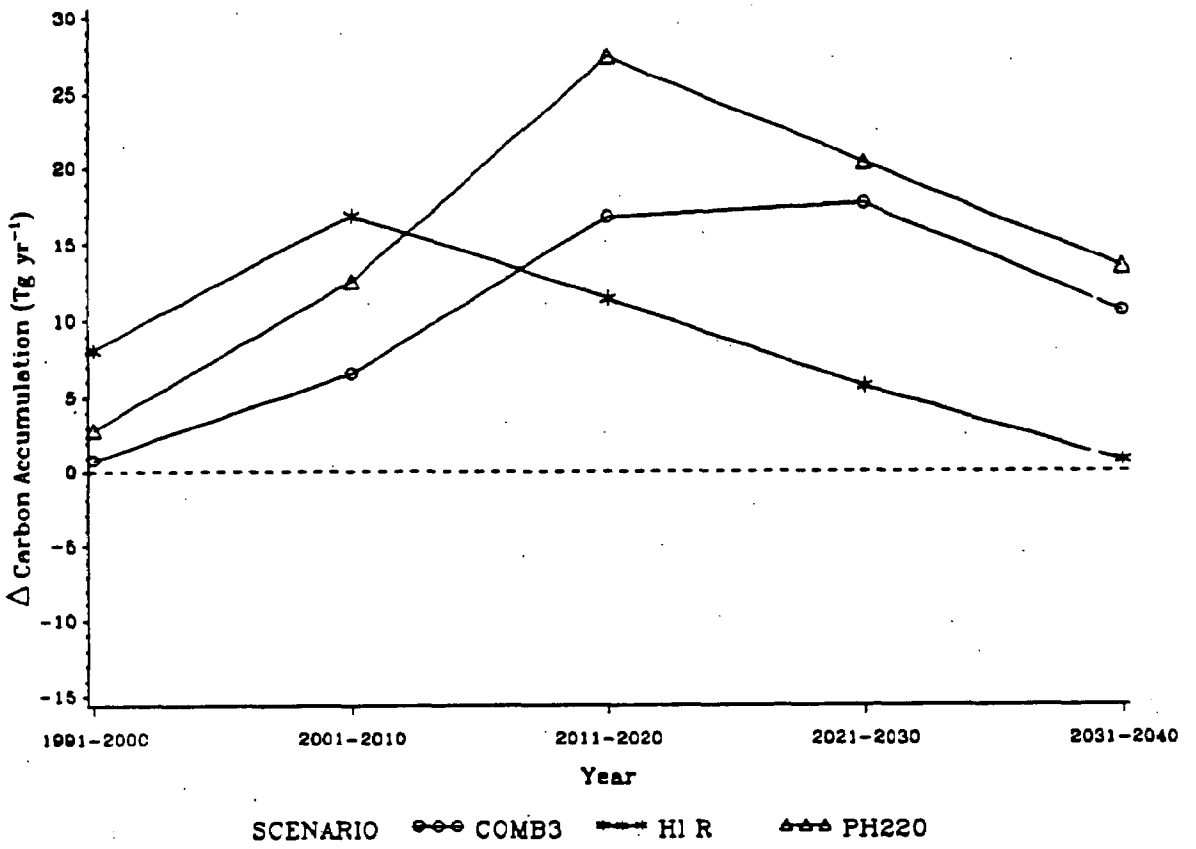

Figure 4.24. Scenario comparison based on the difference from the Current Forest Plans scenario in carbon accumulation rate (Tg-C yr^{-1}). Abbreviations are the same as in Figure 4.22.

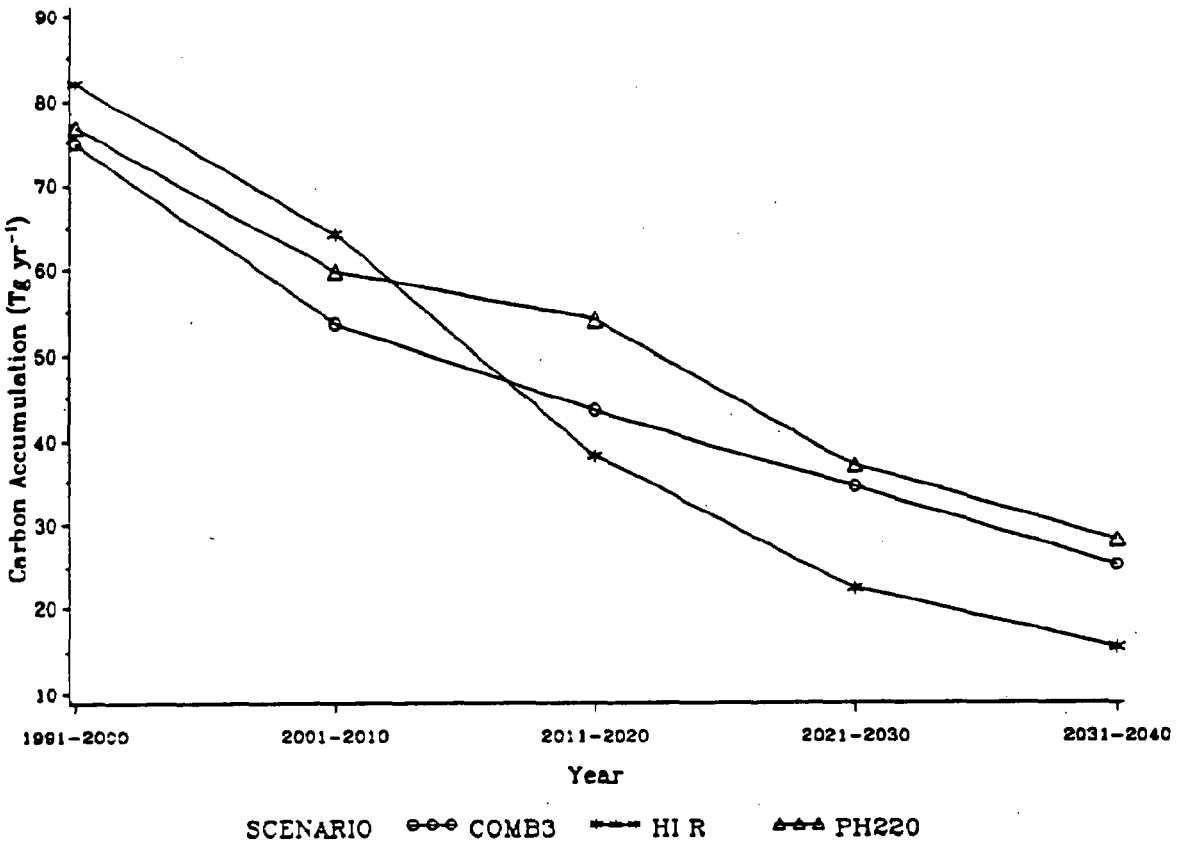

Figure 4.25. Scenario comparison based on carbon accumulation rate (Tg-C yr^{-1}). Abbreviations are same as in Figure 4.22.

results focuses on the time course of the total net flux (i.e. net uptake per year) for the forest land base (Figure 4.25). This perspective emphasizes the magnitude of the responses to policy options compared to the general trend in the forest sink assuming no policy changes.

A complete set of figures showing all scenarios in each of the four formats follows (Figures 4.26 - 4.29).

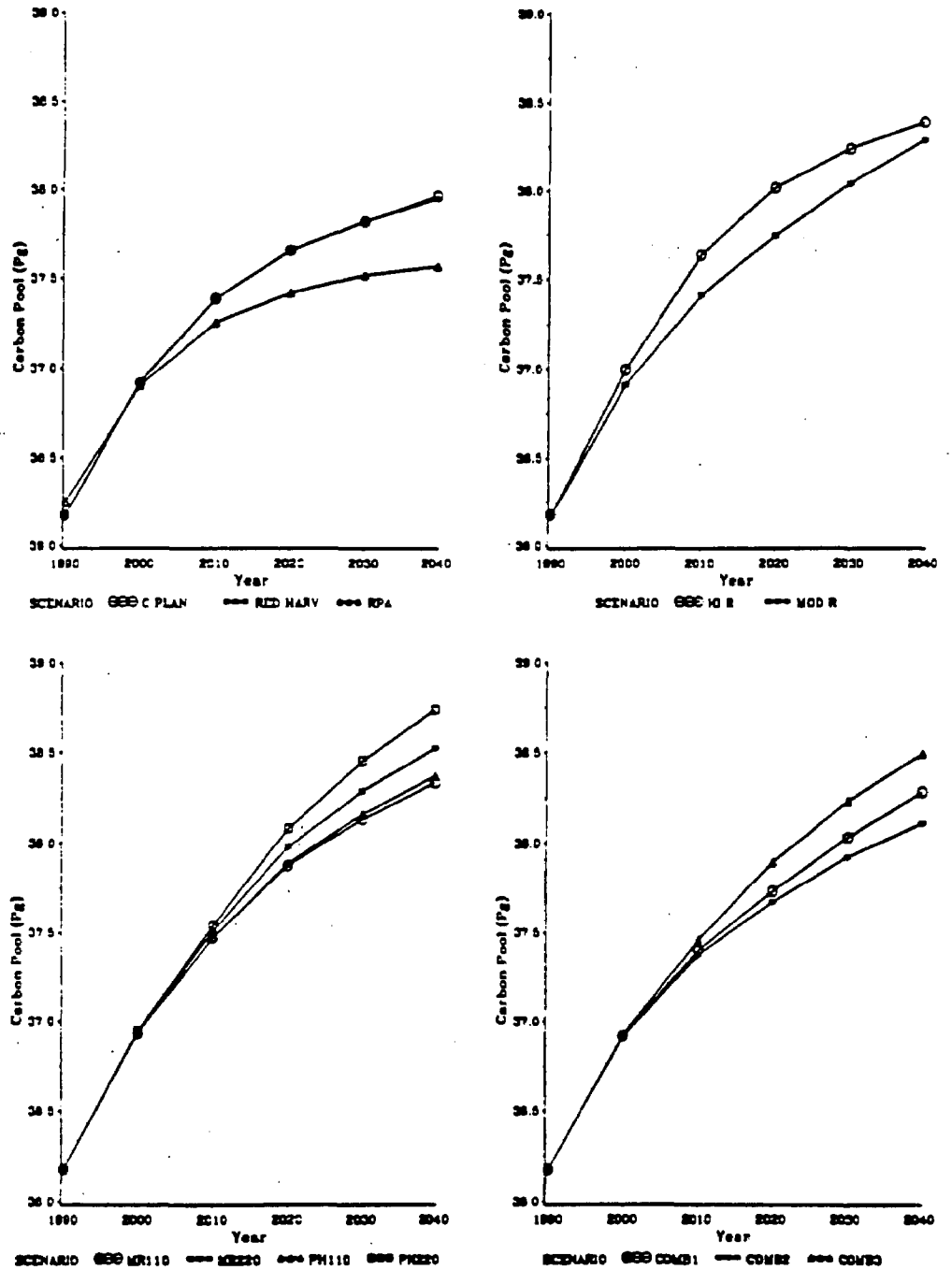

Figure 4.26. Results for all scenarios in the total forest carbon pool (Pg-C) format. Abbreviations are as follows: C Plan = Current Forest Plans, Red Harv = Reduced Harvest scenario, RPA = RPA scenario, HI R = High Recycling scenario, MOD R = Moderate Recycling scenario, MR 110 = Moulton and Richards ($110 million) scenario, MR 220 = Moulton and Richards ($220 million) scenario, PH 110 = Parks and Hardie ($110 million) scenario, PH 220 = Parks and Hardie ($220 million) scenario, COMB 1 = Combination 1 scenario, COMB 2 = Combination 2 scenario, COMB 3 = Combination 3 scenario.

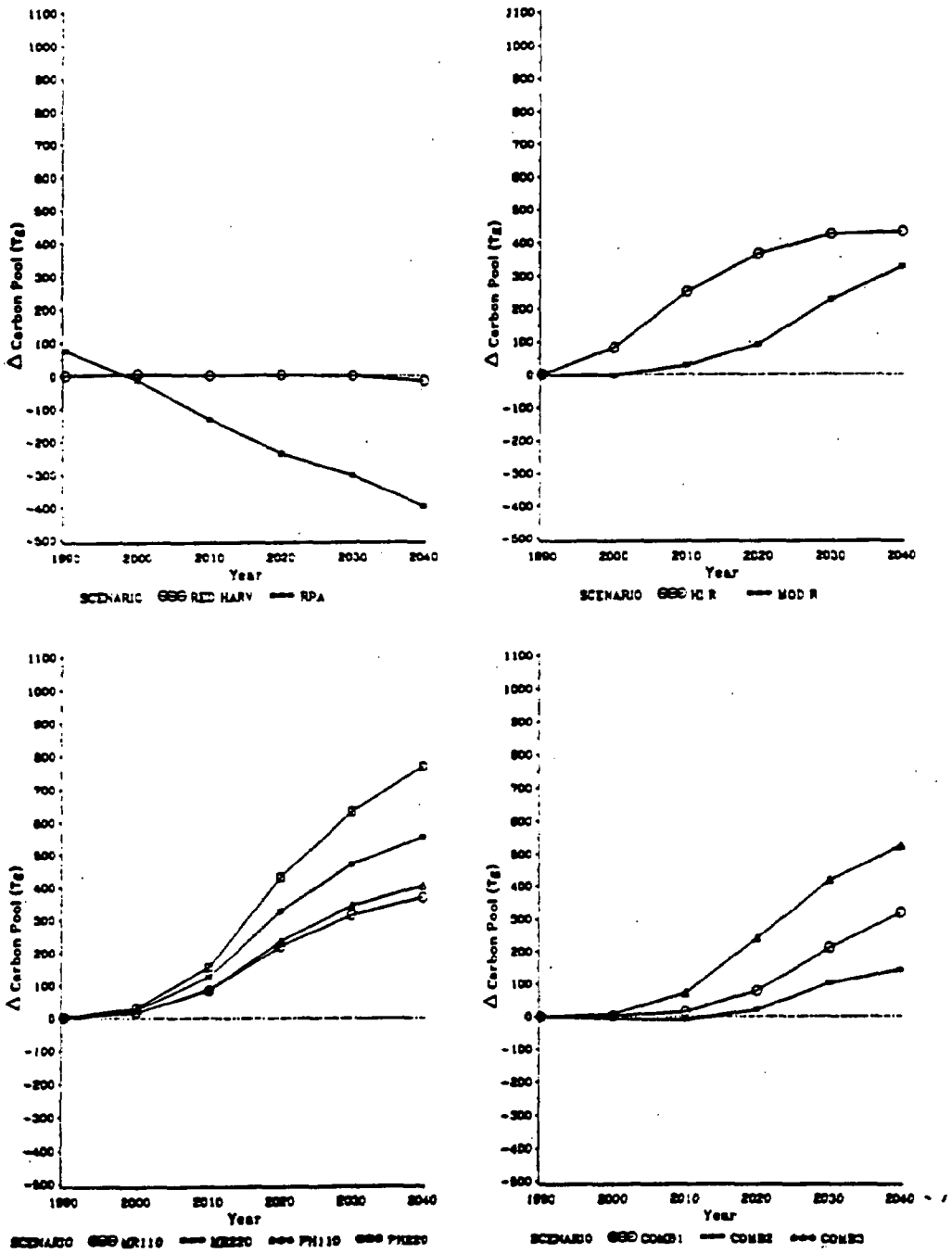

Figure 4.27. Results for all scenarios in terms of differences from the Current Forest Plans scenario in the total forest carbon pool (Tg-C). Abbreviations are the same as in Figure 4.26.

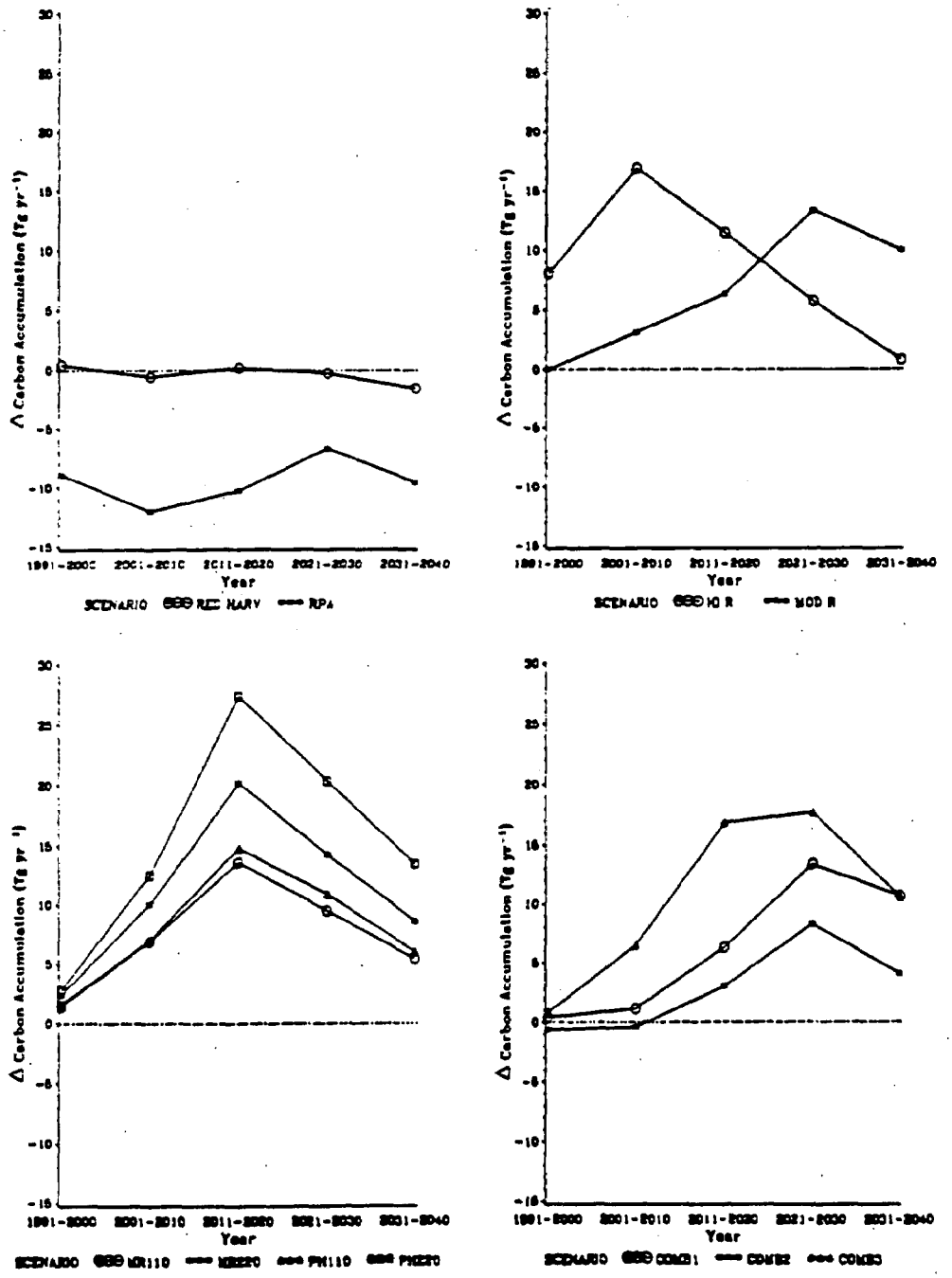

Figure 4.28. Results of all scenarios in terms of the differences from the Current Forest Plans scenario in carbon accumulation rate (Tg-C yr^{-1}). Abbreviations are the same as in Figure 4.26.

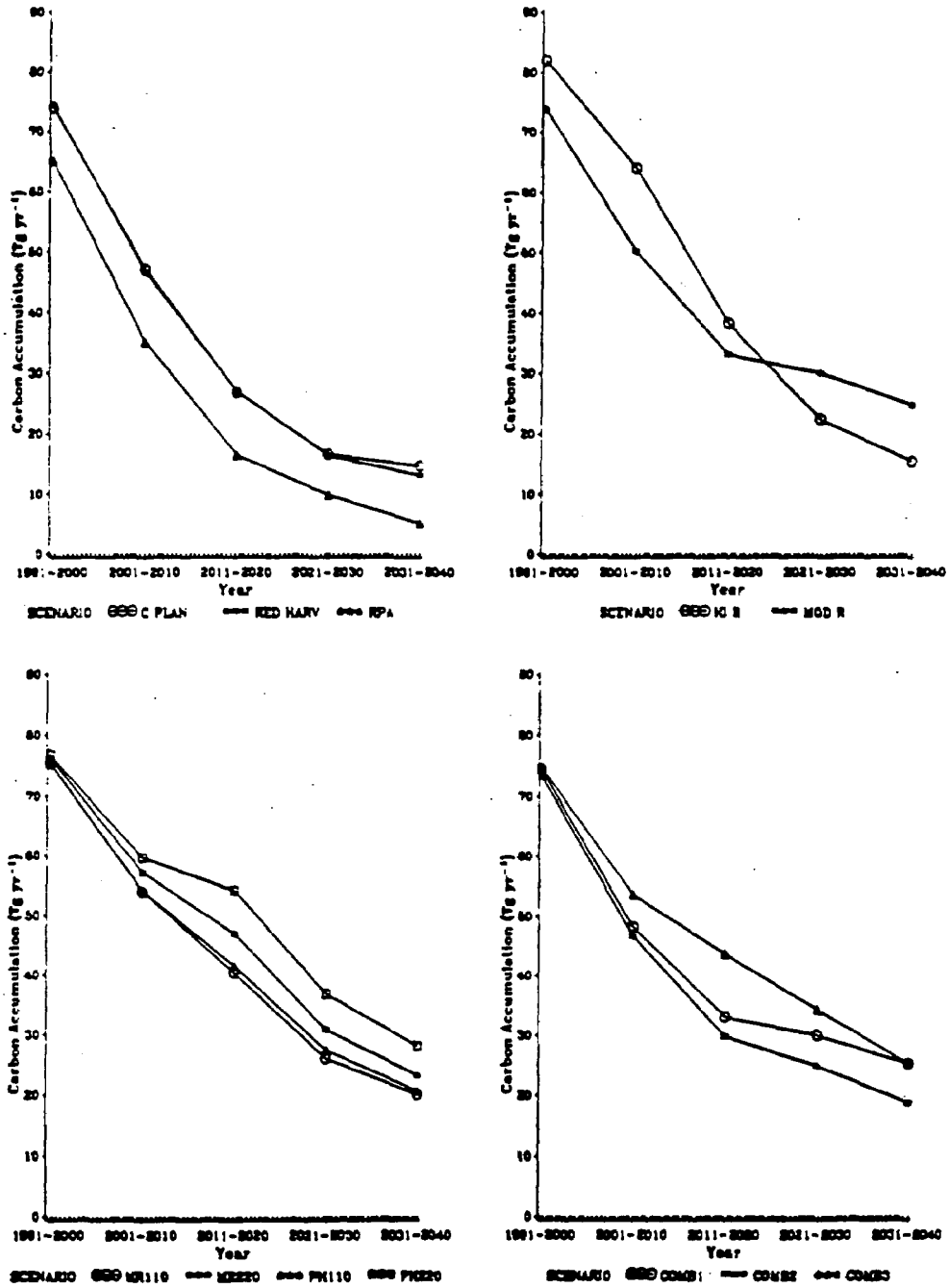

Figure 4.29. Results for all scenarios in terms of the carbon accumulation rate (Tg-C yr^{-1}). Abbreviations are the same as in Figures 4.26.

SECTION 5. POTENTIAL IMPACTS OF CLIMATE CHANGE

Projected climate change could lead to widespread vegetation redistribution. The analyses of carbon budgets earlier in this document ignored possible consequences of climate change. It will be important in future assessments to begin considering climate change impacts. Because of the importance of vegetation in the global carbon cycle, climate changes could reduce or augment terrestrial carbon sources or sinks. These changes could in turn lead to significant feedbacks to the climate system. Here we summarize the potential magnitude of future climate change, the potential response of vegetation to climate change, and the implications of vegetation change for the global carbon cycle.

Climate Change

The effect of a doubling of CO_2 concentrations on global climate has been simulated using General Circulation Models (GCMs). These models are complex mathematical representations of the important processes in the earth system that affect atmospheric circulation and thus climate. Currently the models have significant limitations in their ability to simulate the present climate, especially regional climate patterns (Dickinson 1986, Grotch 1988, Dickinson 1989). Several of the most significant limitations include: (1) the inadequate simulation of cloud processes; (2) the inadequate coupling of atmospheric and oceanic processes; and (3) coarse spatial resolution (Gates 1985, Schlesinger and Mitchell 1985, Dickinson 1986). Nevertheless, GCMs can be used to indicate the potential range of changes in global climate.

All GCMs simulate increases in global mean temperature under doubled atmospheric CO_2 levels, with increases ranging from 1.5 to 4.5° C at a rate of 0.1 to 0.5° C per decade (Table 5.1, Houghton et al. 1990). Global precipitation is simulated to increase from 3 to 15% over current amounts.

Vegetation Change

Several different types of vegetation models have been used to simulate the response of vegetation to climate change. At local scales, forest gap models can be used to simulate changes in tree species composition in response to climate change (Botkin et al. 1972, Shugart and West 1977, Shugart 1984). Forest gap models simulate forest succession after an opening is formed in the forest canopy, caused, for example, by the death of an overstory tree. Currently, the models are primarily based on empirical observations and data on the relative competitive ability of tree species. This limits their usefulness for evaluating the effects of climate change.

To overcome the limitations, efforts are underway to integrate physiological responses into the gap models, for instance, by coupling them with individual tree physiology models such as TREGRO (Weinstein and Beloin 1990, Weinstein et al. 1991). Several different gap models have been used to simulate climate change impacts on forest composition at specific geographic locations (e.g., Urban and Shugart 1989; Botkin et al. 1989; Prentice et al., in press). Solomon (1986) used the gap model FORENA to simulate climate

Table 5.1. Summary of predicted changes in global mean temperature and mean annual precipitation as simulated by four General Circulation Models.

Global Circulation Model	Temperature (°C)	Precipitation (%)
Oregon State University[1] (OSU)	+ 2.8	+ 8
Geophysical Fluid Dynamics Laboratory[2] (GFDL) (R15 and R30 runs)	+ 4.0	+ 8
Goddard Institute for Space Studies[3] (GISS)	+ 4.2	+ 11
United Kingdom Meteorological Office[4] (UKMO)	+ 5.2	+ 15

[1] Schlesinger and Zhao 1989
[2] Manabe and Wetherald 1987
[3] Hansen et al 1988
[4] Wilson and Mitchell 1987

change impacts at 21 sites in eastern North America. His research predicted significant changes in vegetation, indicative of a northward movement of tree species. Subsequent modeling studies in the same region have supported these results (Urban and Shugart 1989, Botkin et al. 1989).

Ecosystem process models that operate at local to regional scales have been developed for grassland and forest systems. Examples include the Forest-BGC model (Running and Coughlan 1988), which simulates ecosystem productivity and hydrologic processes, and CENTURY (Parton et al. 1988), which simulates biogeochemical cycles in grassland ecosystems. These models can be used to simulate changes in ecosystem plant productivity under changing climate conditions. However, these models can not predict changes in the distribution of plant lifeforms or species.

Several different vegetation models have been developed to simulate continental to global scale changes in the distribution of natural vegetation. One class of models is based primarily on correlations of biomes with several climate variables such as temperature and precipitation (Holdridge 1947, 1967; Prentice 1990). A second class of models is based on correlations of the distribution of plant life forms with a suite of climate variables that describe the seasonality of climate (Box 1981). Another class of broad-scale models, including BIOME (Prentice et al. 1992) the Canadian Climate Vegetation Model (CCVM; Lenihan, Oregon State University, personal communication) and Mapped Atmosphere Plant Soil System (MAPSS; Neilson, USDA

Forest Service, personal communication) are based on a mechanistic understanding of the climate factors that have been shown to limit the distribution of plant life forms, such as temperature and soil water. MAPSS is the most sophisticated model in that it includes a mechanistic calculation of water balance in which vegetation cover interactively affects soil moisture (Neilson, USDA Forest Service, personal communication). The model also incorporates a stomatal conductance function which can be modified to simulate the direct effects of enhanced CO_2 concentrations on plant growth.

Before results of the simulation of vegetation redistribution by these different broad-scale models are summarized, it is important to include the direct effects of CO_2 in simulations of future vegetation response. Consider that potential evapotranspiration (PET) would increase significantly under warmer climate conditions (Marks 1990). Increased drought-stress caused by increased evapotranspiration could be the primary cause of vegetation change (Neilson et al. 1989; Neilson 1990). Even though precipitation is predicted to increase globally, the increases would not be sufficient in many regions to offset the increase in evaporative stress on plants caused by the increase in PET. Hence, plants would be subjected to increased drought stress.

However, enhanced concentrations of CO_2 could mitigate the impact of increased drought stress by improving water-use efficiency (Jarvis and McNaughton 1986, Allen 1990, Eamus 1991, Mooney et al. 1991). Water-use efficiency can be defined as the amount of carbon fixed per molecule of water transpired. Under elevated CO_2 concentrations, plant stomates would not need to remain open as long for a specific amount of CO_2 to diffuse into the plant and be fixed, and thus transpirational losses would be reduced. This hypothesized increase in plant water-use efficiency under higher CO_2 concentrations has been demonstrated in greenhouse and field experiments (e.g., Strain and Cure 1985, Drake et al. 1987, Allen 1990). Also, higher CO_2 concentrations have decreased dark respiration in some experiments, which would further increase water-use efficiency (Norby et al. 1992).

Overall, enhanced concentrations of CO_2 increased plant growth rates in many experimental studies (Mooney et al. 1991). However, some short- and long-term studies have demonstrated that the short-term positive growth response may be lost over a period of time, even in the same growing season (Tinus 1972, Tolley and Strain 1984, Surano et al. 1986, Tissue and Oechel 1987, Houpis et al. 1988). Additional, long-term exposure experiments are needed to improve our understanding of the physiological response of plants to higher CO_2 concentrations.

With this background information, scenarios of future vegetation redistribution can be discussed. All the broad-scale models simulate major shifts in vegetation (Table 5.2, 5.3; Neilson et al., USDA Forest Service, personal communication). For instance, the Holdridge life zone model suggests that 16 to 56% of the earth's land surface would change from one vegetation type to another as a result of a doubled-CO_2 climate change (Smith et al. 1992a). Temperate and boreal vegetation tend to shift polewards in these simulations and tropical forests expand.

Predictions of total forest area under doubled-CO_2 climate conditions

Table 5.2. Percent future forest area relative to current forest area using several different vegetation models and climate scenarios. BIOME and Holdridge results reflect global change. CCVM + MAPSS results are for combined areas of US and Canada only.

Vegetation Model	OSU	Global Circulation Model		
		GISS	GFDL	UKMO
BIOME[1]	117	120	119	125
Holdridge[2]	100	101	85	95
CCVM + MAPSS[3] (Current Water Use Efficiency)		47		
CCVM + MAPSS[3] (Increased Water-Use Efficiency)[4]		104		

[1] Prentice et al 1992

[2] Holdridge 1947, 1967, Smith et al 1992a

[3] Neilson, USDA Forest Service, personal communication

[4] Stomatal conductance reduced by 50%.

depend on the vegetation model and climate change scenario used in the simulation (Table 5.2). For the OSU and GISS climate scenarios, the Holdridge life zone model simulates no net change in global forest area, but under the warmer GFDL and UKMO scenarios it simulates a decrease in forested area. The BIOME model simulates a 17 to 25% increase in global forest area. However, the more process-based models CCVM and MAPSS, when used in conjunction, simulate a decrease in forest area in Canada and the US under the relatively mild GISS scenario, assuming no change in current water-use efficiency of ecosystems (Neilson et al., USDA Forest Service, personal communication). If water-use efficiency increases by 50%, CCVM and MAPSS simulate a slight increase in forest area.

In the US, MAPSS simulations suggest that future forest area could range from 2 to 98% of current forest area, depending on the climate scenario, wind speed assumptions, and plant water-use efficiency (Table 5.3). The most extreme decrease in forest area occurs under future doubled-CO_2 climate conditions as simulated by the GFDL model (including simulated future wind speeds) and with current water-use efficiency. Wind speeds are simulated by GCMs to approximately double under doubled-CO_2 climate conditions, increasing PET and thus drought stress. In this scenario, closed forests are restricted to northern Maine and northwest Washington. The least extreme scenario occurs under increased water-use efficiency (stomatal conductance reduced by

Table 5.3. Percent future forest area in the conterminous US relative to current forest area, using MAPSS and several different climate and water-use efficiency scenarios (Neilson, USDA Forest Service, personal communication).

Climate Scenario/MAPSS Simulations	GFDL	GISS
Current Winds and Current WUE[1]	30	5
Current Winds and Increased WUE[2]	98	60
Doubled-CO_2 Winds[3] and Current WUE	11	2
Doubled-CO_2 Winds[3] and Increased WUE[2]	63	16

[1] WUE = Water-use-efficiency.
[2] Stomatal conductance reduced by 50%.
[3] Wind speed conditions for doubled-CO_2 atmosphere predicted by GCM.

50%), current wind speeds, and the GISS simulation of temperature and precipitation.

In summary, these model results suggest that there could be major shifts in the location of forests and other vegetation types under doubled-CO_2 climate conditions. However, the wide range in vegetation change scenarios demonstrates the critical need to: (1) increase our understanding of the direct effects of CO_2 on plant physiology; (2) improve the biophysical foundation of broad-scale vegetation models; and (3) improve the climate models used to simulate doubled-CO_2 climate conditions. The MAPSS simulations demonstrate the importance of including the direct effects of CO_2 in simulating future vegetation response. Lowered rates of stomatal conductance have the effect of reducing drought stress and moderating the resulting vegetation change.

One additional point is that all the above broad-scale models are equilibrium approaches for simulating vegetation response to climate change. That is, they predict the occurrence of vegetation after sufficient time has elapsed to permit its full establishment under the set of new climate conditions. They do not address: (1) transition dynamics, such as seedling dispersion, propagation and survival; (2) the processes of dieback of the previously adapted vegetation; nor (3) the influence of soil properties and nutrient availability. These limitations also need to be addressed in the future development of broad-scale vegetation models.

Effects on Carbon Flux

A change in the distribution of vegetation resulting from climate change could have significant effects on the global carbon cycle, which in turn could affect the rate of climate change. As climate changes, carbon storage in the terrestrial biosphere would change primarily in response to changes in the area of the world's forests and in the age class

distribution of the stands within different forest types (Section 4). If the climate changes, vegetation types characterized by high levels of carbon storage may be replaced by those with lower carbon storage (or vice versa). Hence, a net transfer of carbon between the terrestrial biosphere and atmosphere could occur.

Prentice and Fung (1990) estimated a mass transfer from the atmosphere into above- and below-ground pools of the biosphere of 235 Pg-C of carbon after the world's vegetation has responded to a doubled-CO_2 climate change. Their estimate was based on a simulation of future vegetation distribution using a modified Holdridge model (Prentice 1990) and the GISS GCM run for doubled-CO_2 climate conditions (Table 5.1). The GISS model predicts a warmer world with a relatively large increase in precipitation. Much of the simulated increase in terrestrial carbon storage occurs due to expansion of tropical rain forests. Smith et al. (1992b, in press), using similar techniques, projected increases in terrestrial carbon storage ranging from 8.5 Pg-C to 180 Pg-C. Increases in terrestrial carbon storage of these magnitudes would constitute a significant negative feedback to climate change if the carbon sequestering occurred over a century or shorter time-scale.

Several factors mitigate against the scenario of a strong biospheric negative feedback to climate change. First, human impacts on land use may prevent much expansion of forests, particularly in tropical regions. Second, faster decomposition rates associated with climate warming might cause significant losses of soil carbon (Dixon and Turner 1991). Third, there may be a transient pulse of carbon emissions to the atmosphere generated by dynamics of vegetation redistribution, as described below.

Neilson (1990), King and Neilson (1992), and Neilson et al. (USDA Forest Service, personal communication) developed an approach for exploring the effect of transient vegetation dynamics on carbon emissions. Under doubled-CO_2 conditions, boreal and temperate forests are expected to shift poleward with their warmer and more arid margins dying back and being replaced either by different forest types or by nonforest vegetation. In the intervening areas where lifeform and species associations remain fairly stable, moisture regimes may be expected to vary with changes in PET and precipitation. Carbon would be sequestered from the atmosphere in areas of forest expansion and lost in areas of dieback. In the intervening forested areas where drought stress increases, biomass and carbon loss could occur gradually if the rate of increase of drought stress is slow. If the rate of increase is fast, biomass and carbon loss could occur rapidly. In sum, if vegetation change is slow and gradual there may be minimal effects on terrestrial carbon storage other than a gradual change towards carbon levels simulated for steady state doubled-CO_2 conditions. If vegetation change is rapid and if extensive dieback occurs, there could arise a transitional carbon flux (pulse) to the atmosphere serving as a transient positive feedback to climate change.

Neilson (1990), King and Neilson (1992), and Neilson et al. (USDA Forest Service, personal communication) developed and revised a simple model to provide global estimates of potential carbon fluxes caused by vegetation redistribution. The model, which was constructed using the concepts dis-

cussed above, estimates potential yearly carbon fluxes to the atmosphere caused by forest redistribution. It employs several different estimates of global biogeographic shifts and several different assumptions on the rates and sequences of vegetation change to doubled-CO_2 conditions. The predicted carbon fluxes are primarily dependent on the magnitude of climate change and the resulting vegetation redistribution, the assumed carbon density of existing forests (5 kg m^{-2} or 10 kg m^{-2}), and the time required for the climate to change to the simulated doubled-CO_2 conditions (50 years or 100 years). The model does not factor in changes in tropical forests and only considers changes in above ground carbon storage. The key results of the Neilson et al. (USDA Forest Service, personal communication) study are summarized next.

Consider first the scenario that assumes that forest dieback in response to a changing climate does not commence until 25 years after the climatically-induced expansion of forests into non-forested regions (e.g., the tundra) begins. In this case, carbon would be sequestered at a rate of 0.2 to 1.6 Pg-C yr^{-1} for 25 years, depending upon the rate of climate warming and the assumed above ground biomass density (1 Pg = 1 Gt = 10^{15} g). The faster the rate of climate change (i.e., 50 years) and the higher biomass density (i.e., 10 kg m^{-2}), the greater the sequestration rate. When forest dieback begins, the terrestrial biosphere changes from a net sink to a net source of CO_2. Net emissions of carbon to the atmosphere range from 0.8 to 4.6 Pg-C yr^{-1} for up to 35 years, depending upon the magnitude of the climate-induced vegetation redistribution and the density of carbon. The greater the vegetation change and the higher the above ground density, the greater the emission rate.

If forest dieback is assumed to begin at the same time as forest expansion, the terrestrial biosphere acts as a source of carbon to the atmosphere, with emission rates in the same range as reported above but for a shorter period of time.

In total, forest dieback could result in the emission of 20 to 160 Pg-C of carbon to the atmosphere over 35 years or more. In all the simulations, forest dieback produced a net flux of carbon to the atmosphere. To put these results in context, currently global fossil fuel emissions are 6.0 Pg-C yr^{-1} (Marland et al. 1989, Boden et al. 1992). Vegetation redistribution could produce additional carbon emissions to the atmosphere of over half that amount. However, in almost all the simulations, the carbon released to the atmosphere during dieback was reassimilated by the later regrowth of forests. Thus, the additional carbon added to the atmosphere by forest dieback is transitory: forest regrowth in areas of previous dieback removes the carbon.

Another important point is that these results also show that if forest expansion into non-forested regions occurs before forest dieback begins, the terrestrial biosphere could be a short term but significant sink of carbon, at a rate of 0.2 to 1.6 Pg-C yr^{-1}. This could slow the increase in radiative forcing, but only until climate had changed enough to begin forest dieback. Thus, climate change is likely to cause biospheric changes that could be first a negative feedback to the climate system and then change to a positive feedback to the climate system.

These estimates suggest that the feed-

back potential of biospheric processes are not trivial. Furthermore, biospheric feedbacks are not limited to direct effects of carbon exchange. Other model studies have attempted to delineate the effect of changing vegetative cover on evapotranspiration, ground-level temperature, precipitation, and rates of direct and latent heat flux (Robinson et al. 1983, Rind 1984, Salati 1986, Dickinson et al. 1991). A model analysis of the effect of replacing the forests of the Amazon basin by grasslands showed significant decreases in both evapotranspiration and precipitation (Shukla et al. 1990), suggesting that the hydrologic cycle would undergo significant alteration. Similarly, if the boreal forest were replaced by non-forest vegetation or bare ground, northern latitude temperatures could decrease just through the resulting changes in surface albedo (Bonan et al. 1992).

In summary, the projected changes in global climate and associated changes in the distribution of forests are large enough that climate change must be considered in planning mitigation strategies. For example, if widespread forest plantations are planned, it would be wise to consider where future climate change would permit continued forest growth and where there would be minimal increases in drought stress. Future climate conditions should also be considered in selecting species for planting. For example, it may be appropriate to choose species adapted to warmer and drier conditions than presently occur at a site. Also, since high latitude areas are expected to warm more than low latitude areas and tree density is low in current taiga regions, it may be worthwhile considering forest technologies and management practices that would enable the establishment of forest plantations in these high latitude areas.

For the US, climate change and the resulting vegetation redistribution could have significant effects on the potential for forest management strategies to partially offset US fossil fuel emissions. The most glaring dilemma is posed by the most severe MAPSS vegetation change scenarios in which large fractions of US forests are simulated to dieback during a doubled-CO_2 climate change. The potential exists for forest dieback to change the forest sector from a net sink to a net source of carbon over the next century. Forest management options for increasing carbon sequestration in the US should be developed with this considerable uncertainty in mind.

SECTION 6. FUTURE RESEARCH DIRECTIONS

A. Refine the existing modeling framework and current inventory

Management of public lands played a significant role in three of the policy options explored in this analysis, yet the treatment of the current age class distribution on these lands was highly simplified. Several approaches may yield improved inventory data for input to the TAMM/ATLAS/FCM framework. These approaches include better accumulation of existing inventory data, incorporation of the results of on-going surveys using satellite remote sensing, and initiation of new remote sensing based studies.

The USDA Forest Service FIA units cover public lands in some cases, notably in the Northeast and North Central regions. These data are not readily available but could be collected. There is also a considerable amount of public lands inventory data assembled for individual National Forests as part of their required planning efforts. Such information might be acquired on a piecemeal basis for areas of particular concern.

Several recently initiated surveys using satellite remote sensing have produced age class information on public lands over broad geographic areas. The USDA Forest Service contracted with a private consulting firm to quantify the amount of old growth forest in the National Forests of the Pacific Northwest Douglas-fir region using data from the Thematic Mapper sensor (spatial resolution 30 m) (Tepley and Green 1991). A parallel study was made by the Wilderness Society. Analysis of the same region across all incorporate ownerships classes and with an effort to incorporate additional age classes is being made by Cohen et al. (in press).

These studies will not yield an inventory with 10-year age classes. However, they are likely to provide enough differentiation among age classes to make possible an improved assessment of the regional carbon flux. Additional areas in which satellite remote sensing might be of particular value are Alaska, the Northern Rocky Mountains, and the Central Rocky Mountains. Ultimately, it is desirable to have a national inventory based on satellite remote sensing or other methodology, and to carry such information in inventory models such as ATLAS.

A second major improvement to the current analysis would be an integration of the public lands inventory data into the TAMM/ATLAS/FCM framework. The current version of TAMM assumes that constraints on public land harvests are by forest policy rather than available inventory. As the proportion of public land available for harvest is increasingly restricted, it will become more important to incorporate information on age class distributions, stocking levels, and management intensity on public lands in the TAMM projections. These inventory projections can then be run through FCM to estimate changes in carbon storage.

A more sophisticated treatment of emerging management strategies on public lands (Swanson and Franklin

1992), such as selective cuts and maintenance of tree species diversity would be desirable as public lands were brought into the TAMM/ATLAS/FCM framework. Extensive structural modification of ATLAS might not be required to accommodate these changes.

Additional work is also needed on the dynamics of the woody debris pool and the pool of carbon in forest products and landfills. Better inventory data on woody debris would permit more comprehensive calibration and validation of the modeling approach used for woody debris dynamics in FCM. More information on the fate of harvest residue, particularly on lands converted to non-forest use, would also contribute to development of a balanced woody debris budget. The most important requirement for a thorough treatment of forest products is a time course for the harvest level and a breakout of the harvest into pools with differing residence times as products still in use or still in landfills.

A last area of anticipated improvement is in the user interface for the TAMM/ATLAS/FCM database. Currently, any of more than 400 stand-level carbon budgets can be readily compared using a menu-driven user interface. A second interface allows comparisons of the age class distribution and harvested area by age class within any combination of forest type, region, management intensity policy scenario, or time period. A planned interface will allow an equally flexible examination of the carbon pools and flux by region, time period, and policy scenario.

B. Integrate national-level studies with global scale analyses

As noted in Section 2.A, considerable controversy remains over the question of whether the terrestrial biosphere as a whole is currently a source or sink of carbon, and over the geographic distribution of possible regional sources and sinks. Relatively detailed analyses, such as the present one, may be needed in many countries in order to resolve these questions since land use and management are such strong influences on regional carbon budgets.

The framework for an Agreement on Global Warming, recently signed at UNCED, requires all signers to formulate a national inventory of greenhouse gas emissions For many countries, the forest sector will be a significant component of the national carbon budget. There is a need to combine information from these studies into a common framework, and to compare their results with estimates of current carbon flux (at continental to global scales) derived from alternative modeling strategies. Results of analyses using global carbon cycle tracer models, which estimate carbon flux based on observed spatial and temporal variation in the atmospheric CO_2 concentration, are of particular interest (Keeling et al. 1989, Tans et al. 1990).

C. Evaluate the potential effects of climate change

The analyses in this document, treating how policy changes might impact carbon pools and flux over time periods of decades, are based on the assumption of constancy in the geographic distribution of forest types, in forest productivity and in forest disturbance regimes. To the degree that climate changes in response to increasing concentrations of trace gases, these assumptions may break down. The role of increasing CO_2 per se on plant productivity and on drought tolerance is also likely to impact forest distribution and productivity patterns. The net effect of projected

changes in climate and CO_2 concentration might be to increase or decrease carbon storage on forest lands. Changes in silvicultural practices in response to climate change might also be expected to impact carbon dynamics.

In order to evaluate potential impacts of climate change on the forest sector carbon budget, climate/vegetation models which simulate vegetation response to climate are needed. At the US EPA Environmental Research Laboratory in Corvallis, Oregon, research over the coming years will include development and application of several such models as well as the spatially distributed surfaces for the climate variables which are needed to drive them. These modeling analyses will be made in conjunction with on-going ecophysiological studies which will aid in parameterizing plant responses to multiple environmental variables.

The initial approach to evaluating climate change effects on forest carbon flux will be a regional scale case study of the Columbia River Basin in the Pacific Northwest. Two climate/vegetation models will be spatially distributed over the basin and applied using the current climate and future (e.g., doubled-CO_2) climate scenarios.

The first model, Mapped Atmosphere-Plant Soil System (MAPSS), uses climate drivers and site water balance to estimate potential leaf area index (LAI) and vegetation type (Neilson, personal communication; King and Neilson 1992). MAPSS requires information about soil water holding capacity, and mean monthly estimates of precipitation and potential evapotranspiration. Vegetation type is estimated for each grid cell based on current correlations in the distribution of vegetation and climate variables, particularly the seasonality of precipitation. Potential LAI is likewise based on calibration using observed patterns in LAI and site water balance. MAPSS does not treat photosynthesis, respiration, and nutrient cycling, so a second, more process based biogeochemistry model, will be used to evaluate potential effects of climate change on carbon balance.

One biogeochemistry model being considered is Forest BGC (Running and Coughlan 1988) which was designed for analysis of the carbon, nitrogen, and water cycles at the regional scale. It is a "big leaf" model, meaning that a few simple compartments (including foliage, stems, and roots) are used rather than representations of individual trees. The BGC model estimates photosynthesis, evapotranspiration, and runoff at a daily time step. Carbon allocation to foliage, roots and boles, as well as decomposition of litter and soil is estimated annually. Daily climatic inputs include precipitation, maximum and minimum temperature, dew point temperature and total short wave solar radiation. The site variables include soil water holding capacity, LAI, and the carbon and nitrogen in foliage, stems, roots, litter, and soil. The focus in BGC is on incorporating basic plant physiology in as general a fashion as possible in order to simulate effects of climatic gradients or climatic change on ecosystem mass flux. Comparison of BGC outputs for above-ground net primary productivity and site water balance with observed patterns of carbon and water flux has indicated reasonable agreement for a variety of locations (Running and Coughlan 1988, Running and Nemani 1988, Hunt et al. 1991).

To the degree possible, satellite remote sensing will be used for initializing, calibrating and validating the climate/vegetation models. As an exam-

ple, the recent land cover map based on imagery from the Advanced Very High Resolution Radiometer (AVHRR) sensor (Loveland et al. 1991) provides high spatial resolution (1 km) information on vegetation classification. Additional land surface features potentially detectable by satellite imagery include snow cover, surface temperature, foliar biomass and vegetation structure. The development and deployment of new sensors, on the planned EOS satellites, with higher spatial and spectral resolution will expand the possibilities for coupling models of terrestrial processes to satellite imagery.

SECTION 7. CONCLUSIONS

Terrestrial vegetation is one of the dominant components of the global carbon cycle. An accurate assessment of changes in terrestrial sources and sinks of carbon, particularly as impacted by anthropogenic factors, is necessary to predict future changes in the atmospheric CO_2 concentration.

Forest are of particular importance among terrestrial ecosystems because of their great capacity to accumulate carbon. Because stand age class distributions and forest harvest rates determine the net forest carbon flux, national-level analyses of forest sector carbon budgets, based on forest inventories, are necessary to evaluate terrestrial carbon flux over large areas. Forest sector carbon models also permit evaluation of forest policy options for mitigating the rising level of atmospheric CO_2.

The US forest sector currently (~1990) is a significant carbon sink. Nearly 70% of the accumulation is on the forest land base and the remainder is in forest products still in use or in landfills. The pattern of carbon accumulation and loss among regions, forest types, and ownerships reflects a long-term increase of forest volumes on non-industrial private lands, a relatively young age class distribution on industry lands, and continued transition on public lands from primary forest to a managed condition.

Projections of the forest sector carbon budget indicate a continuous increase in stored forest carbon but a steady decline in the national accumulation rate. The positive rate of sequestration at the national level is the sum of continuous net gains on the public forest land, and a trend of decreasing rates of accumulation on private timberlands. The latter are projected to show net losses of carbon by 2010.

The various policy options treated in this analysis result in reduced harvest levels or increases in the forest land base, which moderate the trend towards lower carbon accumulation rates. The magnitude of the additional carbon sequestered under these options is significant relative to the goal of stabilizing US carbon emission over the next decade.

The effects of projected changes in climate over the coming decades on the US forest sector carbon budget have not been considered in this analysis. Significant alteration of temperature and precipitation regimes may be expected to modify future forest distribution and growth rates, as well as occurrences of wildfire and other disturbances. Coupling of spatially distributed models, which simulate the distribution of vegetation types under different climate scenarios, and process-based ecosystem models, which quantify climatic effects on forest productivity, will be necessary to evaluate these potential climate change effects.

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SECTION 9. SUPPORTING DOCUMENTS

A. Uncertainty in Country-wide Forest Biomass Estimates[1]

Charles E. Peterson[2,3] and David P. Turner[2]

ABSTRACT: Country-wide estimates of forest biomass are the major driver for estimating and understanding carbon pools and flux, a critical component of global change research. Important determinants in making these estimates include the areal extent of forested lands and the associated biomass. Estimates for these parameters may be derived from surface-based data, photo interpretation or satellite remote sensing, with varying degrees of uncertainty. Ground data is typically aggregated by forest type, stand age, productivity level, and ownership. Survey priority is usually given to regions and forest types with timber of commercial value, such that information on understory biomass and forested lands of low commercial value is either absent or of limited reliability. Furthermore, information on below ground biomass, which is costly and time-consuming, is not generally collected. Typically, uncertainty in survey statistics increases as the level of post-stratification increases because of reduced sample size. Likewise, literature-based expansion factors also add to the uncertainty of a final estimate because of the often unknown spatial inference for those factors. Estimates based on modeled processes may provide relatively limited information on uncertainty. The discussions in this paper are based on research funded by the US Environmental Protection Agency, and data provided by the US Forest Service.

Keywords: reliability, inference, sampling error, sensitivity

INTRODUCTION

The major variables requiring consideration in the assessment of uncertainty associated with country-wide carbon budgets include forest land area, the carbon from biomass associated with that area, and the soil carbon. Of these three major areas, we are concentrating here on reliability of biomass estimates insofar as they affect

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carbon pools. It is essential to understand the effectiveness or reliability of the (biomass) estimation procedures if modelers are to manage the expectations that others may have from their interim and final products. Thus, our goal is not to provide absolute values for error in a particular case, but to use some empirical examples to illustrate how one may arrive at estimates of reliability for biomass as the major component in determining carbon pools and flux. Although biomass is the example used for this workshop, the ideas are easily extended to other components of the carbon budgeting process.

As part of the US Environmental Protection Agency's Climate Change Program, we are addressing the area within the USA that is bounded by the conterminous "lower 48" states. In each state, thousands of photo points from current aerial photographs are used to stratify and compute expansion factors for the field (i.e., located on the ground) sample data (e.g., see Bassett and Oswald 1983). The effectiveness of the estimation procedure for area and volume, that is, how representative the estimates are for the real population, is primarily a function of two important measures: precision (sampling error) and the confidence level. An additional component in variation called nonsampling errors (e.g., mistakes in choosing a design, data processing, analytical mistakes, etc.) are usually not known, so we rely on the sampling precision and associated confidence (probability) level to indicate reliability for the survey estimates (Husch et al. 1982). We also recognize that there are many other sources of uncertainty, such as photo/map interpretations, biomass equations, volume equations, and measurement error. Because most of these kinds of uncertainty are not quantified along with their published results, we decided to select the examples used in this paper.

For a US carbon budget, we are using volume and biomass estimates from USDA Forest Service inventory and resource data that are summarized regionally and are derived from a statistically-based survey. Because a major objective is to provide carbon budget information by state, and by types/classes according to ownership, species, and productivity, it is necessary to examine the possible tradeoffs in reliability of estimates when stratifying for those kinds of information. Volume is used instead of biomass as an example, because (i) volume is inventoried nationwide (biomass is not) and (ii) problems of reliability are equally applicable to both measures.

APPROACH

Precision refers to the variation among repeated sample estimates (e.g., the clustering of observations about the average), and is usually expressed as a standard error of the mean (average) of those estimates. Confidence intervals (CIs), expressed by the sample mean ± one or two standard errors, are computed for state resource reports using data sampled from US forested timberlands. Although the confidence intervals vary according to the magnitude of the estimate and variation in the attribute being measured, they provide some approximate reliability for the reported statistics. In this paper, we offer examples of volume and area estimates accompanied by 67% CIs, where the odds are two out of three (67% probability) that the true value (timberland area or volume) will fall inside the range described by the sample mean ± 1 standard error. Although confidence intervals vary with both the size of the estimate and the variance of the item being sampled, we have included examples from data published

in numerous state reports to illustrate how the width of the confidence band changes with the size of the estimate.

In the procedure for estimating biomass over large regions, we generally have data or information that come from three primary sources:

(i) Survey data -- usually statistically based (i.e., has sampling error) and therefore provides some estimate of regional precision;

(ii) Research plots -- results summarized in the literature and hopefully providing some biological understanding but probably no estimates of regional precision;

(iii) Modeling -- may be statistical with some measure of precision, or for understanding or incorporating processes having no precision estimates and perhaps limited generality.

It is important to keep these three sources of information in mind, since they are generally used in combination with each other. As you move from survey information to literature and finally to modeling, you generally lose the ability to track or estimate precision. As an example, our biomass estimates for timberlands are a combination of survey (aboveground biomass), literature values (below-ground biomass, understory, and forest floor), and modeling efforts (e.g., woody debris).

RESULTS

For the examples of reliability presented here, we are using percent sampling error, defined as one standard error of the mean divided by the average (mean) value. In the Eastern US, the percent sampling errors associated with individual state timberland volume estimates are quite low (Figure 9.A.1). The situation in the West is quite similar (Figure 9.A.2), although both the averages and ranges of percent sampling error associated with timberland volume were greater than in the East. In general, for most state reports that displayed 67% CIs, the percent sampling error was around ±5% for areas by major owner groups and when timber volumes were lumped as softwoods or hardwoods (e.g., Lloyd et al., 1986). However, a state-level sampling error of ±1.4% might increase up to ±50% when volume is stratified by forest type (FIA Staff 1985).

Further examples of percent sampling errors taken from data reported in Gedney et al. (1989) and Bassett and Oswald (1983), are given in Figures 9.A.3 and 9.A.4, to illustrate how reliability changes with the size of the estimate. Note that smaller volume estimates are associated with greater uncertainty (Figure 9.A.3), and in the case of area (Figure 9.A.4), the reliability differs as to whether the data is partitioned by owner or by type, with a trend of increased sampling error by type when estimating strata of smaller forest land area.

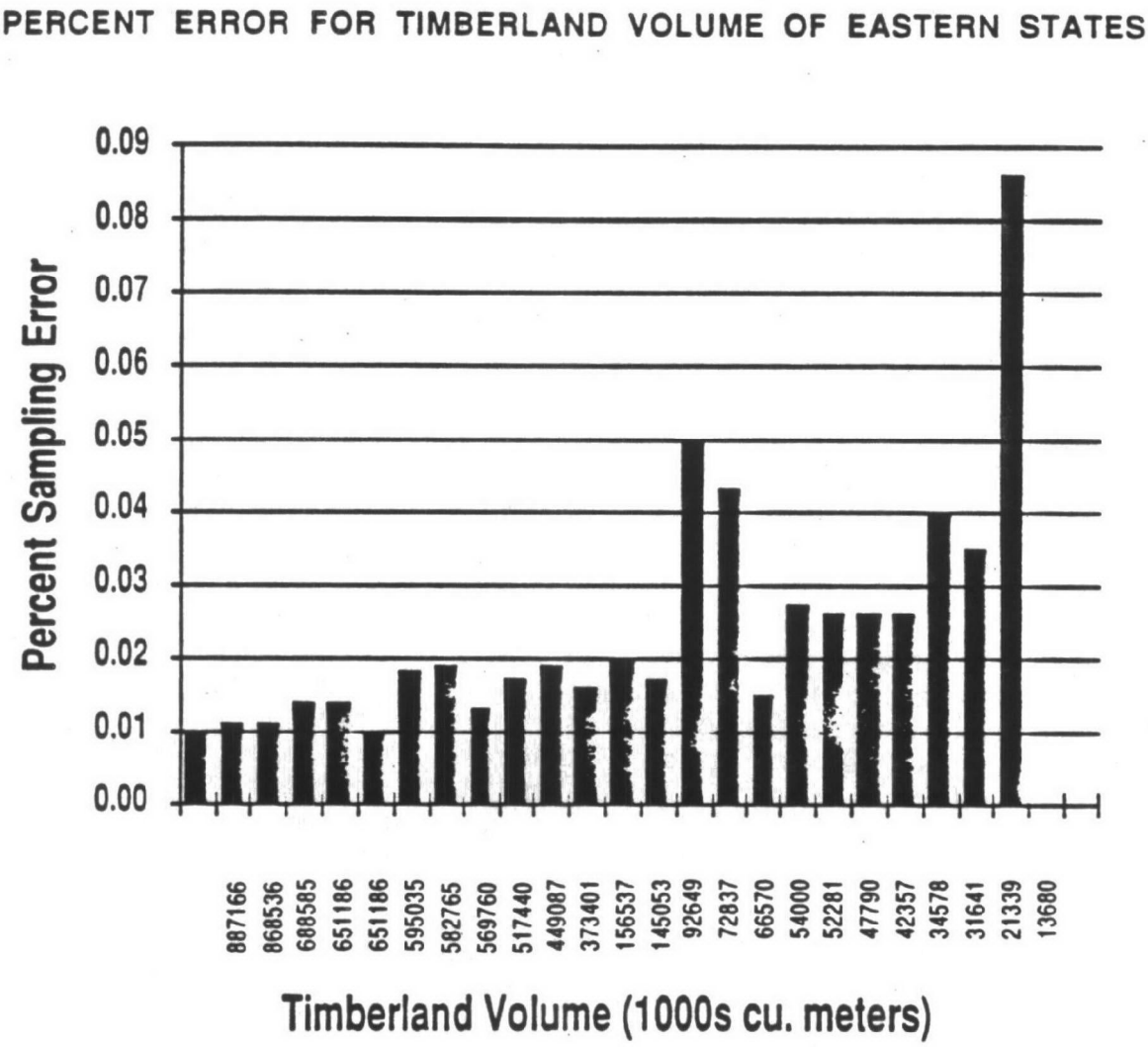

Figure 9.A.1. Percent error (1 s.e. divided by the average value) associated with state level and sub-state level timberland volume estimates the Eastern US.

PERCENT ERROR FOR TIMBERLAND VOLUME OF WESTERN STATES

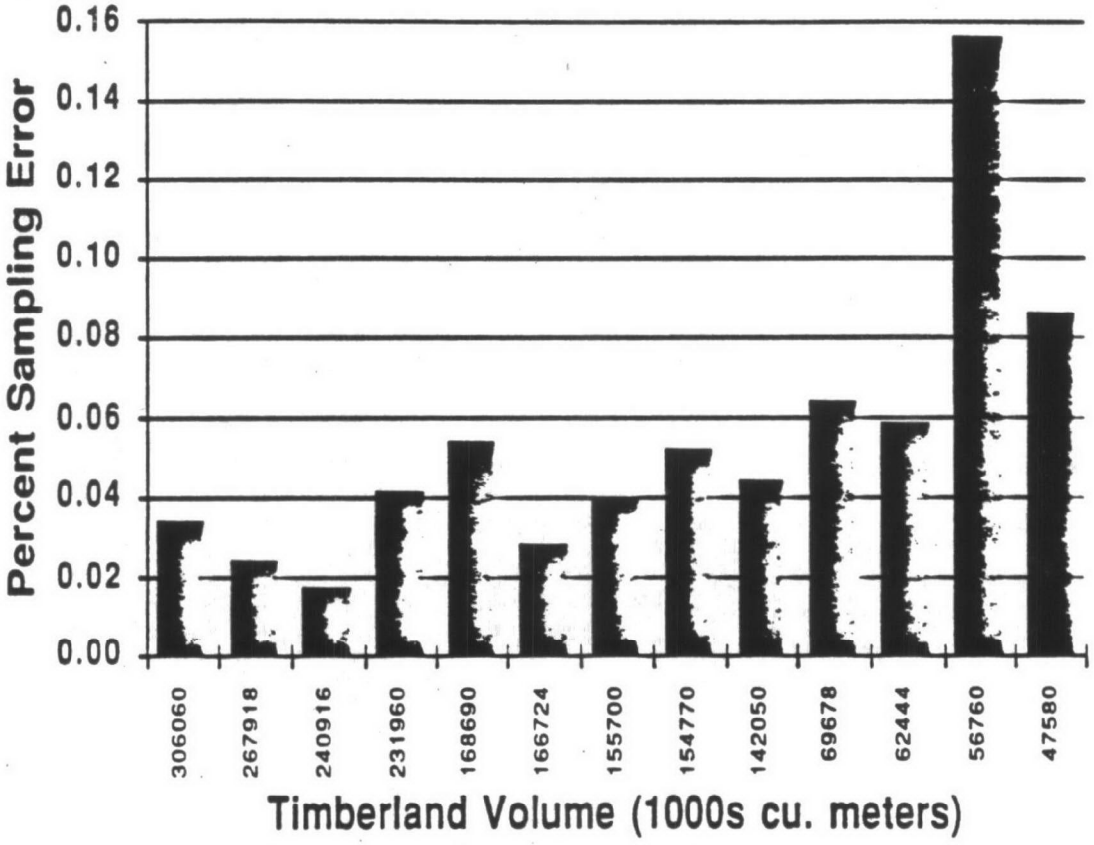

Figure 9.A.2. Percent error (1 s.e. divided by the average value) associated with state level and sub-state level timberland volume estimates in the Western US.

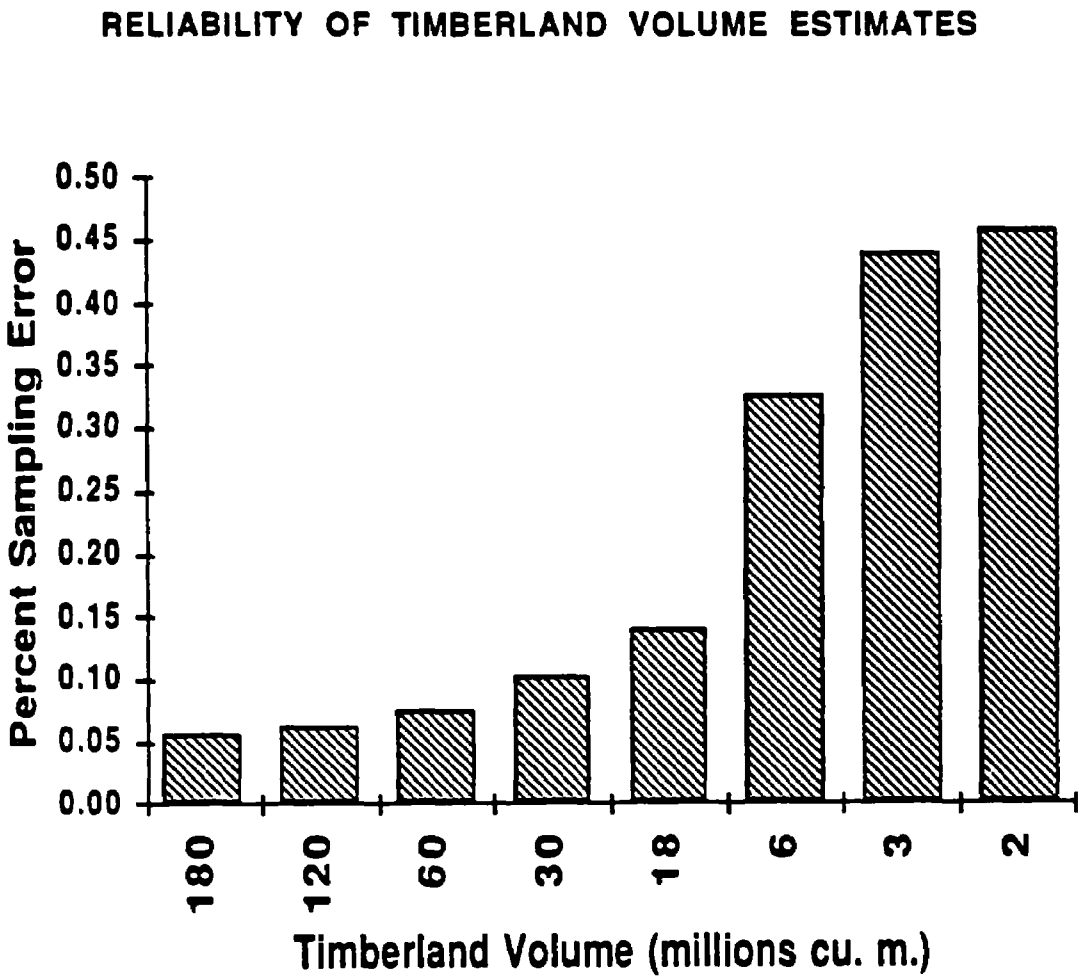

Figure 9.A.3. Percent error (1 s.e. divided by the average value) associated with timberland volume estimates within a state.

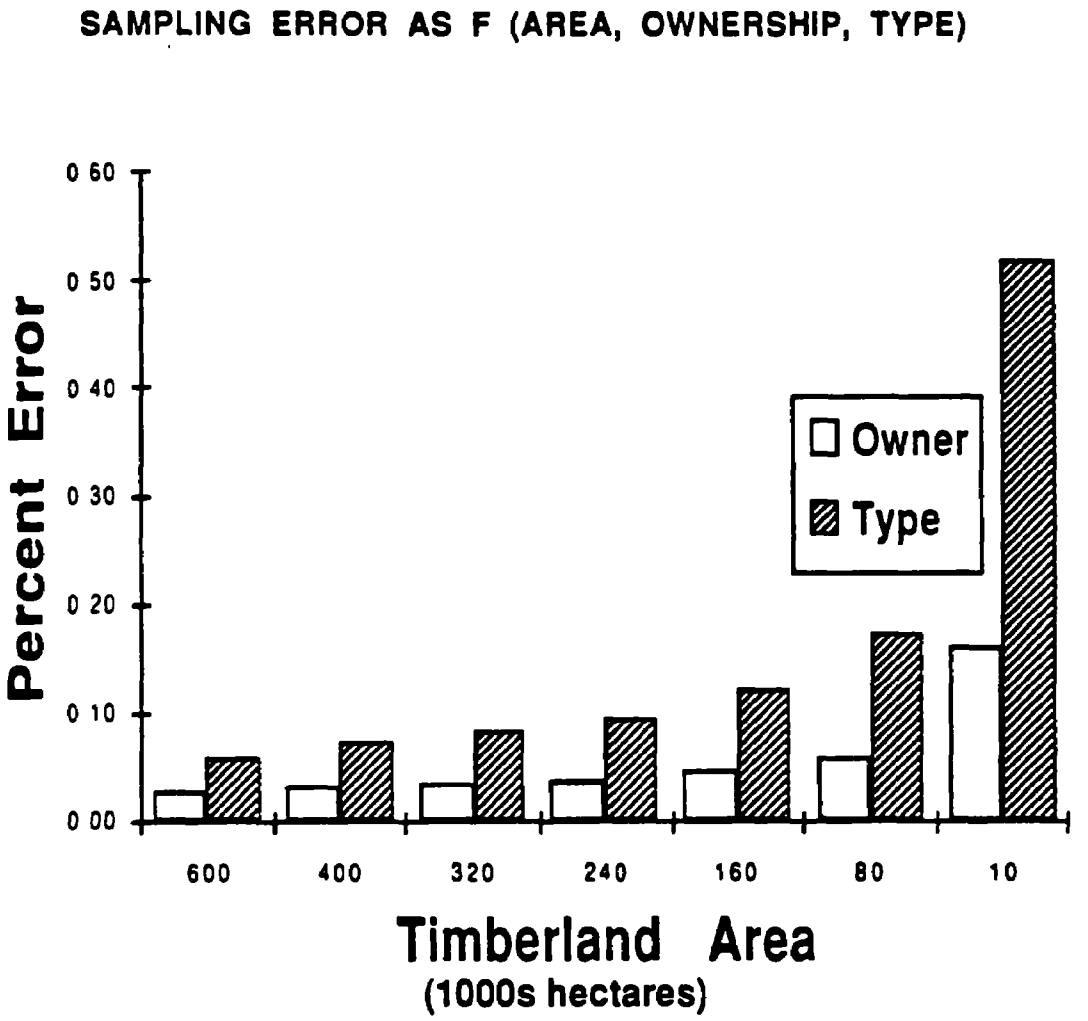

Figure 9.A.4. Percent error (1 s.e. divided by the average value) associated with timberland area estimates stratified by ownership and type within a state.

DISCUSSION

In the above examples, we have illustrated how the sampling errors associated with forest area and volume estimates from statistically-based surveys are relatively small (i.e., high reliability) for large areas. However, uncertainty increases substantially if volume or area are stratified by forest type (the same result is expected for stratifying by productivity class). Because stratification results in a reduced sampling intensity for some combinations of species type or class, an accompanying decreased confidence in the final estimate is expected.

In addition to biomass estimates, most approaches to estimating country-wide carbon budgets will also incorporate results of site-specific studies from the literature and model outputs, neither of which can be characterized with the same kind of reliability that might be expected from a statistically-based survey. A good sensitivity analysis of the models over a range of input values for a variety of scenarios should be employed to assess the reliability of the estimation framework. In order for the end results to be assigned any degree of reliability, even if it is only qualitative, the data origins (i.e., measured or simulated), all assumptions or judgments, and sensitivity analyses, must be fully documented.

Finally, a significant portion of aggregated forest data may not have known reliability. One is then constrained to assuming reliability for that stratum and thus assuming reliability for the aggregate total. In the US for example, volume by age class for all ownership strata has been summarized for the Eastern US. However, similar estimates for public lands in the Western US are not yet summarized, and may require some assumptions about their age-class distributions.

RECOMMENDATIONS

It is important to have some idea as to how our estimation procedures might mask or influence our ability to detect an incremental change in the carbon budget. To that end, we offer the following considerations for reducing uncertainty in country-wide forest biomass estimates:

- Define (with glossary) all components identified with the estimation process.
- Examine the proportional contributions of these components, as well as others (e.g., soils) to the carbon budget.
- Provide some quantitative ranking (importance) of the carbon components according to their magnitude or direction, and how that ranking might differ under climate change scenarios.
- Assign some reliability to the estimates associated with each component, drawing a clear distinction between sources such as statistical sampling error, quantitative sensitivity analyses, and "expert opinion."
- Assess the change in reliability when different spatial scales or finer classifications are employed.

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B. Biomass and Carbon for Woodlands in the Western US

Charles E. Peterson[1,2]

The following table represents a compilation of total tree biomass for woodlands (growth < 1.4 m^3 ha^{-1} yr^{-1}) from most of the western conterminous United States. References corresponding to the citation numbers given in the table represent USDA Forest Service state inventories and resource reports that were used to compile the data. The number corresponds to all computations in the box for a given state. All data are on file at the US EPA Environmental Research Laboratory in Corvallis, Oregon.

In the western conterminous states, excluding Washington and Oregon (no data available), the average above-ground carbon pool is estimated at 1.15 kg m^{-2} (11.5 tons per ha). Below-ground estimates (from Koch 1989) raise those estimates to about 1.3 kg m^{-2}. The average carbon density was quite variable, ranging from 0.5 to 2.1 kg m^2 for Juniper in Wyoming and Colorado respectively, and estimates ranging from 1.0 to 4.0 kg m^{-2} (average of 2.5 kg m^{-2}) for chaparral in California. Average carbon for hardwoods within California, ranged from 0.7 to 1.9 kg m^{-2} (average is 1.6 kg m^{-2}).

Although forest woodlands comprise nearly half of the forested lands in the conterminous Western US, they account for less than 5% of the above-ground carbon pool for ALL forest lands of the conterminous US. While this information may indicate that non-merchantable forests play a small role in assessing carbon pools, these levels of carbon might be a benchmark or indication of the final carbon pools that might be expected if timberlands, under climate change, were to transition to carbon storage and productivity of those lands now classified as woodlands.

Another exception to the contribution of woodlands to the total carbon pool is in Alaska, where by virtue of the extremely large area of land classified as woodland, the total carbon above-ground is greater for these forest lands (about 0.57 Pg-C) than the total carbon pool from Alaskan timberlands (about 0.50 Pg-C).

If the average carbon storage for the woodlands of the western US is applied to the area of woodlands in the conterminous US, then total woodland tree carbon storage is 0.49 Pg-C.

The following references correspond to the citation reference numbers given in Table 9.B.1. These references represent USDA Forest Service state inventories and resource reports that were used to compile total tree biomass for woodlands (growth < 1.4 m^{-3} ha yr^{-1}) in the western conterminous US. The number corresponds to all computations in the box for a given state. All data are on file at the US EPA Environmental Research Laboratory in Corvallis, Oregon.

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Table 9 B 1 Computation of carbon for woodlands in the western US

Where Total bole weight = function of live bole volume, dead bole volume and wood density factor
Total carbon is average carbon density (kg ha^{-1}) from unreserved areas multiplied times total woodland area (reserved plus unreserved)

State	Forest Type	Area-unr 1000s acres	Area-res 1000s acres	Total Area 1000s hectares	Livebole 1000s cubic ft	Deadbole 1000s cubic ft	Density[1a] Factor kg cf^{-1}	Total bole Weight kg10E 06	Top/branch[1b] Weight kg10E 06	Unr C kg10E 06	Unr C kg ha^{-1}	Total Carbon kg10E 06	Total Carbon kg m^{-2}	Total Carbon tons ha^{-1}
WY	P-J	2a 14 3	2a 0 3	6	2b 5068	2c 1094	14 07	86 7	2d 0 02	45 2	7807	46 1	0 78	7 8
	J	691	3 4	281	216127	31292	11 1	2746 4	4 82	1433 4	5126	1440 4	0 51	5 1
	Oak	27 7		11	16582	1658	11 23	204 8	0 13	100 6	8977	100 6	0 90	9 0
	Other	44 7		18	76073	12737	11 23	997 3	0 36	489 9	27080	489 9	2 71	27 1
	SUBTOT	777 7	3 7	316	313850	46781		4035 2	5 33	2069 0	6574	2077 0	0 66	6 6
ID	P-J	3a 611		247	3b 298200		14 07	4195 7	3c 3 91	2188 0	8849	2188 0	0 88	8 8
	Hdwd	207		84	113300		11 23	1272 4	1 72	625 6	7468	625 6	0 75	7 5
	SUBTOT	818	0	331	411500	0		5468 0	5 63	2813 6	8499	2813 6	0 85	8 5
NV	P-J	4a 7156	4a	2896	4b 3498881	220065	14 07	52325 6	4b 23 48	27273 9	9418	27273 9	0 94	9 4
	J	1684		682	560305	34080	11 1	6597 7	32 52	3454 3	5069	3454 3	0 51	5 1
	other	145	25	69	58954	7613	11 23	747 5	1 28	390 1	6649	457 4	0 66	6 6
	SUBTOT	8985	25	3646	4118140	261758		59670 8	57 29	31118 3	8558	31185 6	0 86	8 6
UT	P-J	5a 8846	5a 354	3723				0 0		44387 5	5b 12399	46163 8	1 24	12 4
	Aspen	900	3 5	366				0 0		1848 4	5075	1855 6	0 51	5 1
	Cottnwd	67 5	1 5	28				0 0		138 6	5075	141 7	0 51	5 1
	otherHDW	1536 2	32	635				0 0		3155 1	5075	3220 8	0 51	5 1
	FIR-SPR	391 7	9 1	162				0 0		1965 5	12399	2011 1	1 24	12 4
	DF	195 6	6	82				0 0		981 5	12399	1011 6	1 24	12 4
	PP	95 9	1 9	40				0 0		481 2	12399	490 7	1 24	12 4
	LP	148 1	19 3	68				0 0		743 1	12399	840 0	1 24	12 4
	otherSFT	24	1 4	10				0 0		120 4	12399	127 5	1 24	12 4
	SUBTOT	12205	428 7	5113	0	0		0 0	0 00	53821 4	10897	55862 9	1 09	10 9
CO	P-J	6a 4651 6	6a 185 9	1958	6b 383321	47932	14 07	6067 7	6b 4 41	3163 6	4692	9186 0	0 47	4 7
	J	450 8	1 6	183	587193	75445	11 1	7355 3	8 90	3836 7	21645	3962 9	2 16	21 6
	Oak	705 2	21 8	294	116460	11769	11 23	1440 0	3 65	708 8	2614	769 1	0 26	2 6
	Other	13 5	2 8	7	11471	1058	11 23	140 7	0 09	69 1	13138	86 7	1 31	13 1
	SUBTOT	5821 1	212 1	2442	1098445	136202		15003 7	17 05	7778 3	6896	14004 7	0 69	5 7
AZ	P-J	7a 9070	7b 672 6	3943	7c 5273209	7d 1062855	14 07	89148 4	7e 10 87	46452 0	12655	49896 9	1 27	12 7
	J	2109	0	854	688145	148948	11 1	9291 7	37 86	4860 7	5695	4860 7	0 57	5 7
	Oak	889	26 2	370	397200	58264	11 23	5114 9	6 75	2514 7	6990	2588 7	0 70	7 0
	Mesquite	1268	9 1	517	149484	28235	11 23	1995 8	1 40	980 6	1911	987 7	0 19	1 9
	Other	154	1 3	63	15755	4119	11 23	223 2	0 32	109 7	1761	110 7	0 18	1 8
	SUBTOT	13490	709 3	5746	6523793	1302421		105774 0	57 19	54917 8	10059	58444 7	1 01	10 2
MT	P J	8a 0	8b 8 2	3	8c 0	8d		0 0	8e 0 00	0 0		0 0		
	J	71 35	11 0	33	30206	2372	11 1	361 6	0 53	188 7	6534	217 7	0 65	6 5
	Oak	0	0 22	0	0			0 0	0 00	0 0		0 0		
	Other	0	1 24	1	0			0 0	0 00	0 0		0 0		
	SUBTOT	71 35	20 63	37	30206	2372		361 6	0 53	188 7	6534	217 7	0 65	5 6

NM	P-J	9a	7865 2	9b	136 2	3238	9c	5253747	1311275	14 07	92369 9	9e	24 32	48137 4	15123	48970 5	1 51	15 1
	J		596 1		3 2	243		232594	42288	11 1	3051 2		33 61	1607 2	6662	1615 8	0 67	6 7
	Oak		402 0		2 3	164		264549	40346	11 23	3424 0		2 83	1785 4	10974	1795 4	1 10	11 0
	Other		3 5		3 7	3		486	29	11 23	5 8		0 05	3 0	2164	6 3	0 22	2 2
	SUBTOT		8866 7		145 3	3647		5751376	1393938		98850 8		60 81	51533 0	14361	52387 9	1 44	14 4
CA	Chappri	10a	3604		3844	3014					0 0			35806 8	24550	10g 73997 7	2 46	24 6
	W Juniper	10b	1255		30	10c 520		403800		11 1	4482 2			2335 2	4598	2391 0	0 46	4 6
	P-J		1029		58	440		597600		14 07	8408 2			4380 7	10520	4627 6	1 05	10 5
N coast hdwd		10d	545		12	225	10e	766000		11 23	8602 2			4223 7	19150	4316 7	1 91	19 1
N Inter hdwd			687		45	296		347000		11 23	3896 8			1913 3	6882	2038 7	0 69	6 9
Sacrmto hdwd			1608		75	681		776000		11 23	8714 5			4278 8	6575	4478 4	0 66	6 6
C coast hdwd			1570		226	727		1691000		11 23	18989 9			9324 1	14675	10666 2	1 47	14 7
S hdwd			2435		148	1045		1208000		11 23	13565 8			6660 8	6759	7065 7	0 68	6 8
Total			6845		506	2975		4788000	0		53769 2		0 00	26400 7	9530	28565 6	0 95	9 6
Total hdwds (GS*1 45)							10f	6942600	0		77965 4		0 00	38281 0	13819	41420 2	1 38	13 9
	SUBTOT		12733		4438	6949		7944000	0		90855 8		0 00	80803 8	15681	108967 1	1 57	15 7
TOTALS (All states except WA, OR, and AK)			63,767 9		5,982 7	28,228								285,043 6	11,045	325,961 1		
Average																	1 15	11 5

Abbreviations

States

AK	Alaska
AZ	Arizona
CA	California
CO	Colorado
ID	Idaho
MT	Montana
NM	New Mexico
NV	Nevada
OR	Oregon
UT	Utah
WA	Washington
WY	Wyoming

Species

Cottnwod	cottonwood
DF	Douglas-fir
FIR SPR	fir Spruce
Hdwd	hardwoods
J	juniper
LP	lodgepole pine
otherHDW	other hardwoods
otherSFT	other softwoods
P J	pinyon juniper
PP	ponderosa pine

Other

Area-unr	Unreserved area
Area res	Reserved area
Unr C	Carbon for unreserved lands
N coast hdwd	North coast hardwoods
N Inter hdwd	North interior hardwoods
Sacrmto hdwd	Sacramento hardwoods
C coast hdwd	Central coast hardwoods
S hdwd	Southern hardwoods
SUBTOT	Subtotal

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C. Woody Debris Budgets for Selected Forest Types In the US

Mark Harmon[1]

INTRODUCTION

One of the most important questions facing ecologists today is the role of the biota in the global carbon cycle. The fact that 1 to 3 Pg yr^{-1} of fossil fuel carbon can not be documented (Post et al. 1990) raises serious questions concerning our current understanding of this cycle. Recent efforts to decrease uncertainty of global carbon cycling processes have focused upon improved estimates of land-use change in the tropics (Dale et al. 1991), sequestration in soil humus (Schlesinger 1977, 1990), increased production associated with CO_2 fertilization (Keeling et al. 1989), and sequestration by temperate forests recovering from past harvest (Tans et al. 1990).

While these efforts have been crucial to increase understanding, there may be another fundamental problem with current carbon budgets: dead trees and associated woody detritus are not considered (Kobak 1988, Houghton and Woodwell 1989, Post et al. 1990). This may seem like an unbelievable statement given that >80% of the globe's living biomass (boles, branches, and coarse roots) is woody (Woodwell et al. 1978), yet a quick examination of estimated above-ground detrital fluxes and stores confirms this major oversight (Harmon and Hua 1991). There are many explanations of this development (Harmon et al. 1986). First, decay of woody detritus is a long-term process and, as with other long-term ecological processes (i.e., forest succession), there are few long-term observational studies (Franklin et al. 1990). Second, inappropriate methodology has been used to study dead wood dynamics. For example, litter production is often estimated from a sample area <10 m^2 for one to two years, and although this is suitable for fine litter, it is inappropriate for woody detritus which requires many hectare-years for reliable estimates (Sollins 1982). Finally, there are few estimates of dead tree stores because they have little commercial value and until recently were associated with "unhealthy" forests or bad management practices (Franklin 1989). Thus, little empirical data exists upon which to base estimates or ground assumptions. The time has come to critically evaluate these commonly held assumptions and to begin to include woody detritus in carbon budgeting efforts.

A tacit assumption in many carbon budgets appears to be that tree mortality is a minor flux. This may stem from a common misunderstanding that above-ground litterfall measurements include measurements of tree mortality. While this may seem like a valid assumption, only 40 of the 204 litter production studies cited by Vogt et al. (1986) included fine woody production, and <10 included tree mortality. In the few mature ecosystems that tree mortality has been examined, it is a major pathway of above-ground detrital input, accounting for 30% of the input in old-growth deciduous forests (Tritton 1980), 55% in old-growth conifer forests (Sollins 1982), and 30 to 50% in humid tropical forests (Brown and Lugo, in review). Recent global estimates of tree

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Table 9.C.1. Regression statistics for the relationship between diameter at breast height, stump height, and stump diameter.

Species	B[a]	r^2	N	Diameter Range[b] (cm)
Abies amabalis	0.204	0.926	62	25-102
Acer macrophylla	0.167	0.959	34	25-63
Alnus rubra	0.168	0.854	27	25-51
Pinus monticola	0.217	0.935	45	25-76
Picea sitchensis	0.323	0.968	81	25-140
Pseudotsuga mensiezii	0.183	0.982	89	25-140
Thuja plicata	0.290	0.949	85	25-140
Tsuga heterophylla	0.195	0.935	84	25-140

a The regression equation was in the form of D=B*DBH*SB where D is the difference between diameter at breast height and stump diameter, DBH is the diameter at breast height, SB is the difference between stump height and breast height.

b Diameter of stumps was determined for each 12.5 cm at 7 stump heights of 0.15, 0 30, 0 46, 0 61, 0.76, 0 91, 1.06, 1.21, and 1 37 m.

mortality indicate 3 to 15 Pg-C yr^{-1} of woody detritus is produced within intact stands, while natural catastrophic disturbances add another 1 to 2 Pg yr^{-1} globally (Harmon et al. 1991; Harmon et al., in press). To correctly estimate global above-ground detrital production, total tree mortality input of 4 to 17 Pg-C yr^{-1} must be added to the 28 Pg-C yr^{-1} input by fine litterfall (Meentemeyer et al. 1982). Therefore, a flux at least as large as fossil fuel burning has been excluded from global carbon budgets.

If the decay dynamics of woody and leafy detritus were similar, then a 10 to 40% underestimate of above-ground detritus production might lead to a similar sized underestimate of above-ground detrital stores. In reality, decay rates of woody detritus are an order of magnitude less than leafy litter (Harmon et al. 1986), and given a global input of 4 to 17 Pg-C yr^{-1}, the result would be a potential steady-state store of 85 to 295 Pg of woody detritus carbon (including dead boles, branches, and coarse roots) (Harmon et al., in press). Therefore, a detrital pool at least as large as 70-84 Pg of fine litter carbon (Kobak 1988, Post et al. 1990) has been ignored.

Another common assumption justifying the exclusion of woody detritus is that this detrital pool is in steady-state. From the small amount of data on this component, this appears to be a completely untenable assumption. Given an order of magnitude

difference in decay rate, it implies that woody litter will take an order of magnitude longer to reach steady-state than fine litter. Yet many carbon budgets include successional changes in fine litter stores but exclude woody litter.

In addition to slower decay rates, the period required to reach maximum input rates following disturbance is considerably longer for woody than leafy litter. Fine litterfall peaks at crown closure, which may occur in five years in deciduous forests (Bormann and Likens 1979) and <50 years in coniferous forests (Gessell and Turner 1976). In contrast, the highest rates of woody detrital inputs are associated with disturbances, although in many successional studies this is overlooked. Disturbances are followed by a long period in which woody litter inputs remain low. Input rates then increase as the later stages of succession are reached, a stage which may take >100 years (Harcombe et al. 1990). This implies that stores of woody detritus will not reach steady-state levels for at least a century following disturbance. While many other ecosystem components reach a minimum immediately following disturbance, woody detrital stores are usually highest following natural disturbance, comprising the majority of above-ground mass (85%) and reaching levels as high as 1300 Mg ha^{-1} (Agee and Huff 1987). Woody detritus production rates are usually lowest immediately following disturbance, and increase gradually until the later stages of succession are reached (Tritton 1980, Harcombe et al. 1990). The pulse of woody detritus input associated with disturbance, followed by temporal lag of woody detritus produced by the new stand, typically causes a U- or J-shaped pattern of woody detrital stores during succession (Harmon et al. 1986, Spies et al. 1988, Gore and Patterson 1986). This often means that woody detrital stores are lowest during the middle stages of succession, a time when many other ecosystem parameters are approaching steady-state.

In natural forests, the large mass of carbon and nutrients in woody detritus contributes to ecosystem stability by building soils, providing habitat, and storing nutrients (Harmon et al. 1986). In old-growth conifer forests, woody detritus comprises up to 22% of the above-ground mass and 81% of the above-ground detritus (Sollins et al. 1980). In old-growth deciduous forests, dead wood can also comprise a substantial fraction of above-ground mass (14%) and detrital mass (48%) (Tritton 1980). Even some tropical ecosystems can store a substantial amount of organic matter in the form of woody detritus (20-40 Mg ha^{-1}, Kauffman et al. 1988, Uhl and Kauffman 1990, Whigham et al. 1991).

Woody detritus mass remains quite high throughout succession in natural forests, but in managed forests it is dramatically reduced (Spies et al. 1988). In managed forests, woody detritus is removed by timber harvest, salvage operations, and firewood gathering (Cramer 1974). This may represent one of the most dramatic, long-term and widespread departures from the natural forest resulting from management (Harmon et al. 1990). This is because other ecosystem components recover more quickly than woody detritus, which does not increase until large trees are present in stands (Spies et al. 1988). Given this long recovery time, short-term rotations can reduce ecosystem carbon storage of woody detritus even when the living portion of the ecosystem has recovered (Harmon et al. 1990).

Excluding dead trees has several profound impacts upon understanding forest carbon dynamics. First, the size of the terrestrial detrital pool has been underestimated (Harmon and Hua 1991, Harmon et al., in press). If woody detritus was in steady-state this might not be a major concern; however, land-use changes have placed woody detritus in disequilibrium. In the Pacific Northwest, for example, reduction of woody detritus accounted for 33% of the carbon flux to the atmosphere associated with the conversion of old-growth forests to plantations (Harmon et al. 1990). Is this result unique to this region, or does it apply to other forests in the boreal, temperate, and tropical regions? If this is a general rule, then past calculations of carbon flux from forest clearing have been underestimated in past studies (i.e., Melillo et al. 1988). It also implies that a large, unaccounted carbon sink may be present in forests recovering from past harvest (Harmon and Hua 1991). Second, excluding dead trees will lead to an unrealistic assessment of the response of detrital stores to projected climate change.

Analysis of transient responses to projected climate change indicate a large pulse of carbon may be injected into the atmosphere as forests are disturbed or migrate (Neilson and King 1991). To a large degree, the temporal dynamics of this transient pulse will be controlled by the decomposition of woody plants killed by catastrophic disturbances (i.e., fire) or increased stress. Yet our understanding of decomposition and mortality rates is so poor that rates have to be guessed for major forest regions (e.g., taiga forest). Third, the estimated effect of intensive forest management on carbon sequestration (Sedjo and Solomon 1991) would probably be reconsidered if woody detritus could be taken into account. Removal of dead wood from forest ecosystems by current sustained, intensive silvicultural practices may represent one of the most dramatic, long-term and widespread departures from the natural forest which result from management (Harmon et al. 1990).

In this report I provide preliminary estimates of the input rate, decay rate, and steady-state stores for major forest types in the United States. For the purposes of this analysis I have included all large and small dead wood that occurs above- and below-ground. Thus my estimates include dead tree boles, stumps, tops, branches, and coarse roots. Unfortunately, there is not a wealth of data to draw upon. However, enough is known to place reasonable limits upon these rates and stores. At this point, the error made in excluding woody detritus from budgets is probably greater than the errors associated with poor parameter estimates.

METHODOLOGY

Mortality Turnover Rates

Mortality turnover rates of woody detritus were estimated from USDA Forest Service forest survey data. Resource Bulletins for states containing a given forest type were used to derive a minimum and maximum ratio of growing stock mortality to growing stock volume. While an extensive computer database exists, I only had access to the published summary reports. This meant a number of assumptions had to be made to make estimates of woody inputs from the forest survey reports. First, I assumed that most woody inputs are associated with tree death. As there are few reliable estimates of branch or coarse root mortality available, there was no other alternative to this

assumption. Therefore, branch inputs and dead coarse root inputs are likely to be underestimated. Second, mortality data were reported for all forests, regardless of age. Therefore I assumed the estimated turnover rates would apply to all ages of forest of a given type. In reality mortality turnover rates (as opposed to absolute input rates) are likely to decrease with forest age. Third, the mortality and volume data are reported for species and not by forest type. I therefore assumed that the majority of each species grew in its respective forest type. This third assumption would probably have little effect on the overall estimates of turnover.

Finally, I assumed that the turnover rate of non-growing stock was similar to growing stock. This is also likely to lead to an underestimate of the overall turnover rate since smaller suppressed trees, or those with extensive defects, are more likely to die than trees considered growing stock. However, this error may not be particularly large for some forest types. For example, in the Southeastern Region merchantable volume statistics are given for both growing stock and total live trees (Bechtold et al. 1987a, 1987b, Thompson 1989). For southeastern conifer forests the mean ratio of total living merchantable volume and growing stock merchantable volume was 1.01 (standard error = 0.02, N=18), indicating non-growing stock volume is a minor component of stands. For southeastern hardwood forests the proportion of non-growing stock merchantable volume increases, with a mean total live to growing stock ratio of 1.14 (standard error = 0.02, N=18), indicating the possibility of a larger underestimate of stand level mortality turnover rates. Given that three of the four assumptions made concerning mortality will cause underestimates of the turnover rates, the mortality turnover estimates should be considered conservative.

The fraction of the tree bole left on the site in tops decreases with increasing tree and decreasing top diameter (Keays 1971). To estimate the volume left in conifer tops, I assumed conifers were cones, therefore the fraction of bole volume left in tops (Tf) was approximately:

$$Tf=(Dtop/DBH)^3 \quad [4]$$

where DBH is the diameter at breast height, and Dtop is the top diameter. For the purposes of this analysis I assumed the top diameter was 10 cm for all conifer forests. In the case of hardwood species, I assumed that top mass would be included in the estimates of branch mass.

The amount of decay in stems is a function of species, location (Boyce 1932), tree age and size (Boyce 1932, Englerth 1942). As trees become older and larger the proportion of the stem with rot increases. These complex relationships were not modeled, instead I used standard volume deductions (Boyce 1932, British Columbia Forest Service 1966, Englerth 1942, Hepting 1971) to estimate the bole mass left as decayed or cull wood.

Breakage of boles during felling also can convert merchantable bole volume into pieces too small to be yarded. Breakage differs widely between species (British Columbia Forest Service 1966), and increases with tree size and roughness of topography (Boyce 1932). I increased breakage with diameter, as observed for Douglas-fir (Boyce 1932), to estimate the mass left in broken conifer boles. For

hardwoods I assumed the majority of breakage would occur in the branches and therefore made no breakage deductions for these species.

Non-merchantable woody portions of trees left after harvest were derived from published and unpublished biomass equations (Grier and Logan 1977, Sanantonio et al. 1977, Tritton and Hornbeck 1982). All branches were added to the woody detrital pool, whereas roots were only added if the dominant tree species in a forest were not able to sprout. All coniferous forests, except those dominated by redwoods, were assumed to have complete root mortality. In the case of redwood forests I assumed that 20% of the root systems would die. In the case of oak-hickory, maple-beech, and bottom land hardwoods, I assumed the root systems would all survive harvest. In the case of oak-pine, I assumed that 50% of the roots would die following harvest. For hardwood forests containing birch, I assumed 33% of the roots would die following harvest.

To estimate the amount of woody detritus left on the site after harvest, I applied these relationships to the published diameter distributions of major US forest types. Diameter distributions were taken from stand tables for Douglas-fir (McArdle and Meyer 1930), western hemlock-sitka spruce (Meyer 1938), ponderosa pine (Meyer 1937), lodgepole pine (Haig 1932; Moir et al., in review), northern hardwoods (Reed et al. 1986), southern pines (Anonymous 1929), spruce-fir (Moir et al., in review), and yellow poplar (McGee and Della-Bianca 1967). For each forest type, I estimated the woody mass left on the site at rotation age under three scenarios: (1) all above-ground mass removed, (2) all boles removed, and (3) only sound, unbroken boles removed. In forest types without readily available stand tables, I used the proportions from the most similar forest type as a substitute.

Decay Rates

Estimates of woody decay rates were derived from the literature and unpublished data. As the majority of literature reports values for woody debris (>10 cm diameter), and the majority of woody material is in the coarse fraction, I used woody debris decay rates as the starting point to determine overall decay rates. These values were then increased based on data from studies with branch, and/or root decay rates (Abbot and Crossely 1982, Cline et al. 1981, Erickson et al. 1985, Fahey et al. 1988, Harris et al 1972, Miller 1983, Yavitt and Fahey 1982). To make this adjustment, I assumed that 15 and 20% of the total dead wood input was in the form of branches and roots, respectively. I then calculated a weighted average decay rate for the forests in which the decay rates of root, branch and coarse wood rates were known. In the case where only root or branch decay rates were known, I assumed branch and root decay rates were the same.

As there were a number of forest types and regions for which no data on woody decay rates existed, it was necessary to extrapolate from known forest types and regions. The effect of climate upon woody decay rates is poorly understood. Increasing temperature may lead to an increase in decay rate, although much of this effect appears to be an interaction with moisture content. For coarse woody detritus, excess moisture tends to limit decomposition more that excessive drying. Increasing temperatures increase the rate of drying, which in turn increases the rate of decay.

Rather than try to model these rather complex interactions, I increased the decay rate 33% if the extrapolation was southward and decreased the decay rate 33% if the extrapolation was northward. In the case of mixed species stands, I used the rates for the fastest decaying and slowest decaying species to estimate the range of decay rates.

Steady-State Stores

The median, minimum, and maximum steady-state solutions for dead wood pools in old-growth forests were calculated for each forest type. These calculations follow the equations presented in Olson (1963) in which the steady-state mass (Xss) is:

$$Xss = I/k$$

where I is the input rate in Mg ha^{-1} yr^{-1} at steady-state, and k is the annual decay rate.

The minimum steady-state stores were calculated using the maximum decay rate and the minimum mortality turnover rate. The maximum steady-state stores were calculated using the minimum decay rate and the maximum mortality turnover rate. The median steady-state stores were calculated using the median decay rate and the median input turnover rate.

To judge if the calculations were reasonable, I compared the estimated steady-state stores to published and unpublished values. In order to make these comparisons it was necessary to make some adjustments of the published data. One difficulty is that many coarse wood studies only measured downed wood (i.e., logs) and did not include snags. In sites where only log mass was reported, I set the snag mass equal to 0.75 log mass, which is the mean snag/log ratio from Harmon et al. (1986). In the case of redwood forests, where there are probably few snags, I computed a range of total woody detrital mass by correcting for roots and branches and by correcting for roots, branches and snags. Another problem is that no published studies of woody debris includes dead roots or branches leading to an underestimate of total dead wood mass. In order to estimate dead roots and branches from coarse wood estimates, I multiplied by 1.2, assuming that although branches and coarse roots form approximately 35% of the tree mass, the decay rates of these forms of detritus are slightly higher than the coarse wood fraction.

As there are few biomass values to compare with estimates of steady-state stores, I also used the ratio dead wood mass to live wood mass as another check. Harmon and Hua (1991) found that ratios of 0.2 to 0.3 are common for temperate conifer forests. Muller and Liu (1991) found that this ratio for northern hardwoods ranged between 0.13 and 0.19, whereas for southern hardwood forests it ranged between 0.05 and 0.17.

PARAMETER ESTIMATES

Input Rates

The published mortality turnover rates of growing stock differed greatly among forest types, from a low of 0.0003 yr^{-1} for redwoods to a high of 0.027 yr^{-1} for North Central spruce/fir (Table 9.C.2). As might be expected, forest types with extremely long-lived species (e.g., redwoods and Douglas-fir) had the lowest turnover rates, whereas forests with short-lived species (e.g., aspen/birch, southeastern pines) had high turnover rates. Forests with insect outbreaks (e.g., lodgepole pine, spruce/fir) or disease outbreaks (such as bottom land hardwoods in the North Central region which have Dutch elm disease) also had high turnover rates.

Comparison of estimated and observed mass input rate of woody detritus indicates considerable agreement for most forest types with detailed ecological studies (Figure 9.C.1). A region by region comparison indicates that :

1. In the Pacific Northwest estimated inputs of 130 to 308 and 160 to 477 g-C $m^{-2}yr^{-1}$ for Douglas-fir and western hemlock/sitka spruce, respectively, are similar to the 120 to 330 (Sollins 1982) and 210 to 380 g-C $m^{-2}yr^{-1}$ values (Sollins 1982, Harcombe et al. 1990) observed for these types.

2. In the Rocky Mountains, woody inputs were 20 g-C $m^{-2}yr^{-1}$ in an Arizona ponderosa pine forest (Avery et al. 1976), as compared to an estimated input of 16 to 38 g-C $m^{-2}yr^{-1}$. In contrast, estimated inputs for Rocky Mountain spruce/fir and lodgepole pine of 100 to 230 and 60 to 170 g-C $m^{-2}yr^{-1}$, respectively, are considerably higher than the 110 and 25 g-C $m^{-2}yr^{-1}$ observed in these types (Moir et al., in review). Therefore, it is likely that mortality turnover rates, mass inputs, and steady-state stores have been overestimated for these two types (see below). A more reasonable estimate of mass inputs for lodgepole forests results if the mortality turnover observed in the Pacific Northwest is substituted.

3. In the Southeast forest types, estimated inputs of 70 to 103 and 130 to 160 g-C $m^{-2}yr^{-1}$ for the oak/hickory and natural pine forest types, respectively, were close to the observed values of 120 and 150 g-C $m^{-2}yr^{-1}$ (Harris et al. 1973).

4. In the Northeast/North Central regions, the estimates for spruce/fir inputs of 78 to 151 g-C m^{-2} yr^{-1} bracket the observed input of 110 g-C m^{-2} yr^{-1} (Frank and Blum 1978). For white/red pine forests, the estimated range of 86 to 235 g-C m^{-2} yr^{-1} is close to the 80 to 180 g-C m^{-2} yr^{-1} observed (Jensen and Zasada 1977, Peet 1984). In contrast, the estimated range of inputs for maple/beech/birch forests of 50 to 60 g-C m^{-2} yr^{-1} is on the low end of the observed range of 80 to 180 g-C m^{-2} yr^{-1} (Eyre and Longwood 1951, Gosz et al. 1973). This might lead to an underestimate of steady-state stores for this forest type. Finally, the estimate of 30 to 80 g-C m^{-2} yr^{-1} for oak/hickory spans the measured value of 60 g-C m^{-2} yr^{-1} (McMillan 1981).

Table 9.C.2. Rate constants of dead wood input for US forest types based upon the ratio of growing stock mortality to growing stock volume.

Region Ecosystem	Mortality constants (yr^{-1}) minimum	 maximum
Pacific Northwest		
Douglas-fir[4,9]	0.0030	0.0070
Spruce/fir	0.0030	0.0070
Hemlock/spruce[4,8]	0.0030	0.0090
Lodgepole pine[1,5]	0.0020	0.0070
Hardwoods[4,12]	0.0040	0.0080
Ponderosa pine[1,5]	0.0040	0.0060
Redwoods[12]	0.0013	0.0010
adjusted[15]	0.0020	0.0030
Rocky Mountains		
Douglas-fir[3,6]	0.0013	0.0033
Ponderosa pine[3]	0.0014	0.0034
Spruce/fir[3,7]	0.0086	0.0200
adjusted[15]	0.0050	0.0075
Larch2,6	0.0034	0.0042
Lodgepole pine6,7	0.0047	0.0131
adjusted[15]	0.0020	0.0070
South/Southeast		
Oak/hickory[10,14]	0.0062	0.0087
Oak/pine[10,14]	0.0086	0.0108
Bottom land[10,14]	0.0080	0.0082
Natural pine[10,14]	0.0110	0.0129
Planted pine[10,14]	0.0110	0.0129
Northeast		
White/red pine	0.0031	0.0085
Spruce/fir	0.0070	0.0130
Oak/hickory	0.0040	0.0110
Maple/beech/birch	0.0060	0.0072
North Central		
White/red pine[11,13]	0.0031	0.0085
Spruce/fir[11,13]	0.0140	0.0270
adjusted	0.0070	0.0130
Maple/beech[11,13]	0.0035	0.0037
adjusted15	0.0060	0.0720
Aspen/birch[11,13]	0.0146	0.0149
Bottom land[11,13]	0.0128	0.0159
Oak/hickory[11,13]	0.0020	0.0055

1. Bassett and Oswald 1983, 2 Bensen et al 1988, 3 Conner et al. 1990, 4 Gedney et al 1986, 5 Gedney et al 1989, 6. Green et al 1985, 7. Green and Conner 1989, 8 Harcombe et al 1990, 9 Harmon and Hua 1991, 10 Kelley and Sims 1990, 11. Kingsley 1991, 12. Lloyd et al 1986, 13. Spenser et al 1989, 14 Vissage and Duncan, 1990, 15. Value recommended to be used in order to match estimated and observed steady-state stores.

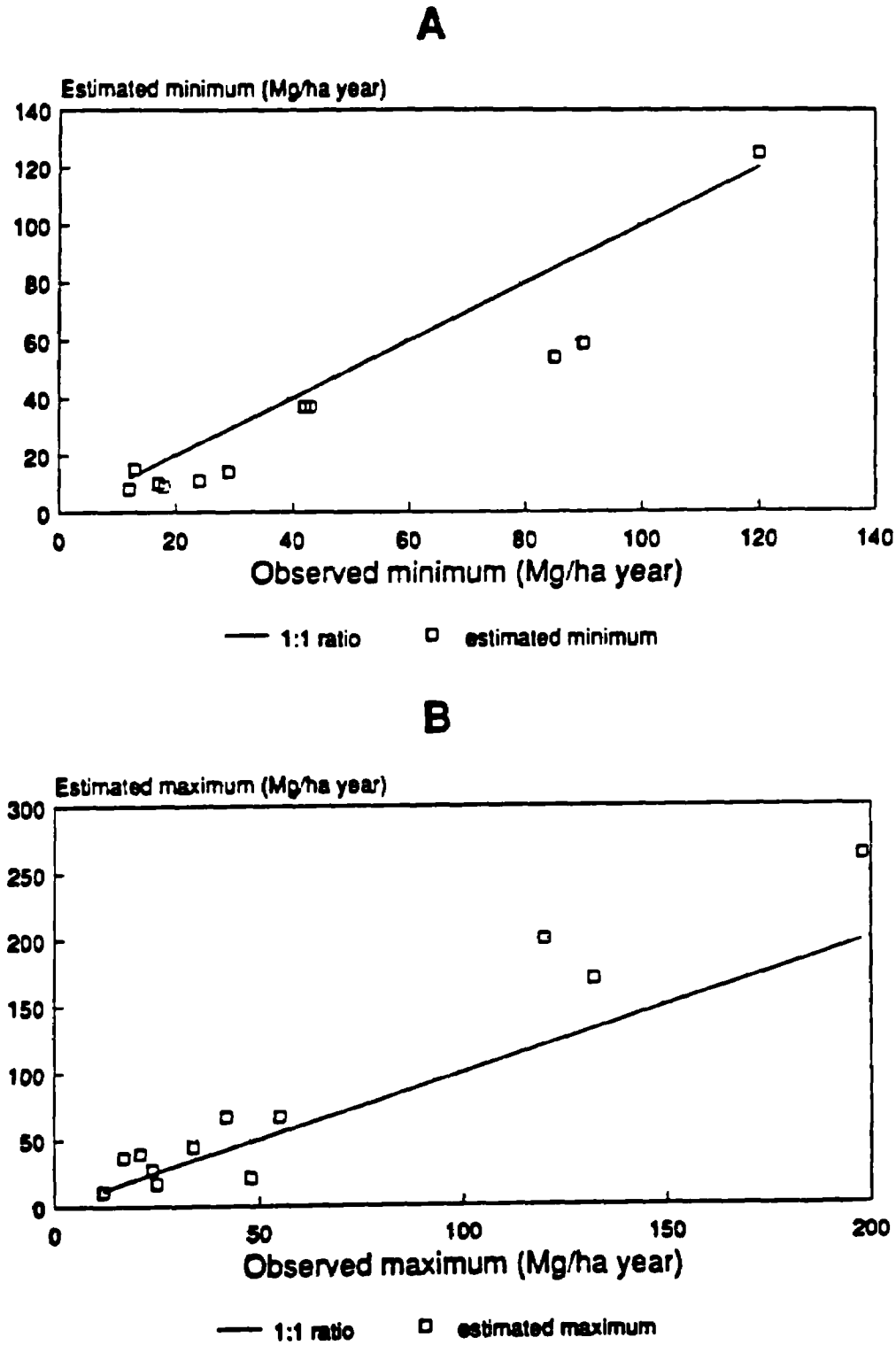

Figure 9.C.1. Observed versus estimated steady-state mass of woody detritus for US mature to old-growth forests. The arrows indicate estimates where one of the parameters was adjusted. A. minimum steady-state mass, B. maximum steady-state mass.

Harvest Input Rates

The estimates of woody detritus left on the site after harvest (including boles, tops, stumps, branches, and coarse roots) did not appear to be highly sensitive to forest age. For example, I examined the proportion of woody detritus left on site as a function of stand age for a Douglas-fir forest with a site index of 3 (McArdle and Myer 1930). At age 60 years, the amount of tree mass left on site for whole tree, whole bole and only merchantable harvests was 13, 34 and 62%, respectively. For a 160-year old stand the estimated residual material for the three harvest levels was exactly the same. This indicates that the intensity of harvest is far more important in determining the mass of woody material left on site than stand age.

In contrast to forest age, the amount of woody material left after harvest was strongly a function of forest type (Table 9.C.3). For a whole tree harvest system, which removes all parts except the roots, the lowest input was for oak-hickory (0% of total biomass) and the highest was for eastern pine forests (22% of total biomass). Total removal of the boles was estimated to leave between 20% (redwood forest) and 54% (aspen-birch) of the total live mass on the site after logging. In a harvest system that does not remove tops, broken or decayed boles, a substantial fraction of the formerly live biomass is left on the site following harvest. These estimates range between 30 and 65% of the total biomass that is left on the site, with the later proportion being typical for non-sprouting forests. The percentage of biomass left on the site after harvest is up to twice the 33% estimated by Houghton et al. (1983) for temperate forests. The rational for the estimates by Houghton et al. were not explained, although they are likely to be best guesses. As my approach demonstrates, it is unlikely that hardwood and conifer forests would have the same proportions, given the ability of the former to root sprout.

The absolute mass added by timber harvest strongly depends upon forest age at the time of harvest. In a Douglas-fir forest, for example, leaving breakage and decayed boles would result in 12.4, 23.2, and 29.7 kg-C m^{-2} at age 55, 105 and 155 years, respectively. Since the stores of previously dead wood in 55 to 155-year old Douglas-fir forests range from 5 to 10 kg-C m^{-2}, a substantial increase will occur during harvest. Moreover, accounting for the decomposition of this detritus indicates a harvested Douglas-fir forest is a net source of carbon to the atmosphere for at least 25 years. This is a period when decomposition of wood detritus exceeds the estimated uptake of tree and understory carbon.

Decay Rates

Decay rates of coarse wood are highly variable among forest types and can vary by an order of magnitude (Table 9.C.4). The lowest decay rate constant was estimated to be 0.005 yr^{-1} for redwood and the highest, 0.08 yr^{-1}, was measured in bottom land hardwood forests of Tennessee (Onega and Eickmeier 1991). While the former value was estimated, it was set equal to a value observed for western redcedar (Harmon, unpublished data), a species that is probably less resistant to decay than redwood. As expected, hardwood forest types generally have higher decay rate constants than conifer forests (Harmon et al. 1986).

Table 9.C.3. The proportion of live tree carbon left on site after timber harvest as woody detritus for US forest types.

Region Ecosystem	Proportion of Live Biomass Left Harvest System Whole Tree	Whole Bole	Merchantable Bole
Pacific Northwest			
Douglas-fir[3]	0.18	0.35	0.56
Spruce/fir	0.18	0.41	0.60
Hemlock/spruce[5]	0.16	0.48	0.66
Lodgepole pine	0.18	0.41	0.60
Hardwoods	0.06	0.52	0.56
Ponderosa pine[6]	0.20	0.45	0.65
Redwoods	0.05	0.20	0.45
Rocky Mountains			
Douglas-fir	0.18	0.35	0.56
Ponderosa pine	0.20	0.45	0.65
Spruce/fir[7]	0.18	0.41	0.60
Larch[2]	0.18	0.35	0.56
Lodgepole pine[7]	0.18	0.41	0.60
South/Southeast			
Oak/hickory[4,8]	0.00	0.28	0.36
Oak/pine	0.11	0.32	0.45
Bottom land[4]	0.00	0.24	0.30
Natural pine[1]	0.22	0.38	0.59
Planted pine[1]	0.22	0.38	0.59
Northeast			
White/red pine	0.22	0.38	0.59
Spruce/fir	0.18	0.41	0.60
Oak/hickory	0.00	0.28	0.36
Maple/beech/birch[8]	0.06	0.52	0.56
North Central			
White/red pine	0.22	0.38	0.59
Spruce/fir	0.18	0.41	0.60
Maple/beech[8]	0.00	0.46	0.50
Apen/birch	0.08	0.54	0.58
Bottom land	0.00	0.24	0.30
Oak/hickory	0.00	0.28	0.36

1. Anonymous 1929, 2. Haig 1932; 3. Mcardle and Meyer 1930; 4. McGee and Della-Bianca 1967; 5. Meyer 1937; 6. Meyer 1938, 7. Moir et al., in review; 8. Reed et al. 1986

Table 9.C.4. Decomposition rate constants for US forest types as estimated from the literature.

Region Ecosystem	Decay rate constants (yr^{-1}) Coarse wood minimum	 maximum	 Adjusted Overall minimum	 maximum
Pacific Northwest				
Douglas-fir[5]	0.011	0.016	0.015	0.022
Spruce/fir	0.015	0.020	0.021	0.028
Hemlock/spruce[3,4]	0.009	0.010	0.013	0.014
adjusted[14]	0.020	0.022	0.028	0.031
Lodgepole pine[1]	0.015	0.020	0.031	0.041
Hardwoods	0.050	0.080	0.052	0.082
Ponderosa pine[6]	0.010	0.012	0.014	0.017
Redwoods	0.005	0.007	0.010	0.014
Rocky Mountains				
Douglas-fir	0.011	0.016	0.015	0.022
Ponderosa pine	0.010	0.012	0.014	0.017
Spruce/fir[11]	0.009	0.011	0.013	0.014
Larch	0.011	0.016	0.015	0.022
Lodgepole pine[1,11]	0.010	0.011	0.021	0.023
South/southeast				
Oak/hickory	0.040	0.050	0.060	0.075
Oak/pine[13]	0.030	0.040	0.045	0.060
Bottom land[12]	0.060	0.080	0.084	0.112
Natural pine[13]	0.020	0.040	0.028	0.056
Planted pine	0.030	0.040	0.042	0.056
Northeast				
White/red pine	0.020	0.030	0.028	0.042
Spruce/fir[2,8]	0.020	0.030	0.028	0.042
Oak/hickory	0.040	0.050	0.060	0.075
Maple/beech/birch	0.050	0.080	0.052	0.062
North Central				
White/red pine	0.020	0.030	0.028	0.042
Spruce/fir	0.020	0.030	0.028	0.042
Maple/beech	0.060	0.080	0.062	0.082
Aspen/birch[10]	0.050	0.080	0.052	0.082
Bottom land	0.060	0.080	0.084	0.112
Oak/hickory[9]	0.030	0.040	0.045	0.060

1. Fahey 1983, 2. Foster and Lang 1982, 3. Graham and Cromack 1982, 4. Grier 1978, 5. Harmon and Hua 1991, 6 Harmon unpublished data, 7. Harmon 1982, 8. Lambert et al. 1980, 9 McMillian 1988, 10 Miller 1983, 11. Moir et al in review, 12. Onega and Eickmeier 1991, 13. Smith and Boring 1990, 14. Recommended changes to make estimated and observed steady-state mass match

Calculations of the factor needed to correct coarse woody decay rates to include all forms of woody detritus indicated a dependence upon forest type. The rate for all forms of woody detritus was 1.40, 2.05, 1.52, 1.41, and 1.03 times the coarse wood rate for Douglas-fir (Cline et al. 1981; Erickson et al. 1985; Harmon, unpublished); lodgepole pine, (Yavitt and Fahey 1982) oak/hickory (Abbot and Crossley 1982), bottom land hardwoods (Harris et al. 1972), and maple/beech/birch (Fahey et al. 1988, Miller 1983), respectively. Given the lack of studies in which coarse wood, roots and branch decay have been compared, these correction factors are extremely tentative. Nonetheless, the overall trend appears reasonable, in that the decay rate of coarse wood is largely determined by the amount of decay resistant heartwood present (and which is largely absent in coarse roots and branchwood). Therefore, one would expect that forest types with trees possessing decay resistant heartwood (Douglas-fir, pine, oak) would have a larger correction factor than those with less resistant trees (i.e., maple, birch).

Using the estimated overall wood decay rate constant indicates that, given a steady input of woody detritus, a steady-state would be reached in 27 to 300 years depending upon the forest type. Using rate constants of 0.02 and 0.08 yr^{-1} (which are more typical of the majority of forest types) would give a steady-state mass within 37 to 150 years. Coupled with the lag in reaching old-growth levels of woody detritus input, this reinforces the notion that woody detrital stores will not be in steady-state unless harvest rotations greatly exceed current practices.

Steady-State Stores

An examination of the original steady-state solutions indicated a number of forests differed widely from reported steady-state stores (Figure 9.C.2, Table 9.C.5). Moreover, in many cases the ratio of dead wood to living biomass was beyond the ranges expected. There were three possible sources of error in these estimates: (1) the decay rate constant, (2) mortality turnover rates, and (3) steady-state live biomass. The third factor seems to explain, in part, why the original estimate of steady-state in redwood forest was low. In most cases, however, the lack of fit was probably caused by a poor estimate of the mortality turnover rate. In the following text each forest type with field data is compared to the estimates and required adjustments are discussed.

Pacific Northwest Forests. The steady-state stores for Douglas-fir forest are within the observed range of 9,000 to 33,000 g-C m^{-2} for this type (Spies et al. 1988, Agee and Huff 1987). The latter value is somewhat unusual for the Douglas-fir region as a whole because it occurred on a very cool, moist site. This would greatly reduce the decay rate and result in a high mass of dead wood. A more typical upper mass would be 12,000 g-C m^{-2} (Harmon et al. 1986). In western hemlock/sitka spruce forest, Grier (1978) reported 13,200 g-C m^{-2} of woody detritus which is much lower than the estimated minimum. My own unpublished data from mature hemlock-spruce stands indicating an average of 8,520 g-C m^{-2} for this type, indicated that a maximum steady-state estimate is unrealistically high. The input rates for this forest type are quite reasonable, suggesting that the decay rate constants have been underestimated in past field studies. I therefore recommend that decay rate constants of 0.028 to 0.031 yr^{-1} be used in modeling dead wood dynamics in this forest type. For redwood

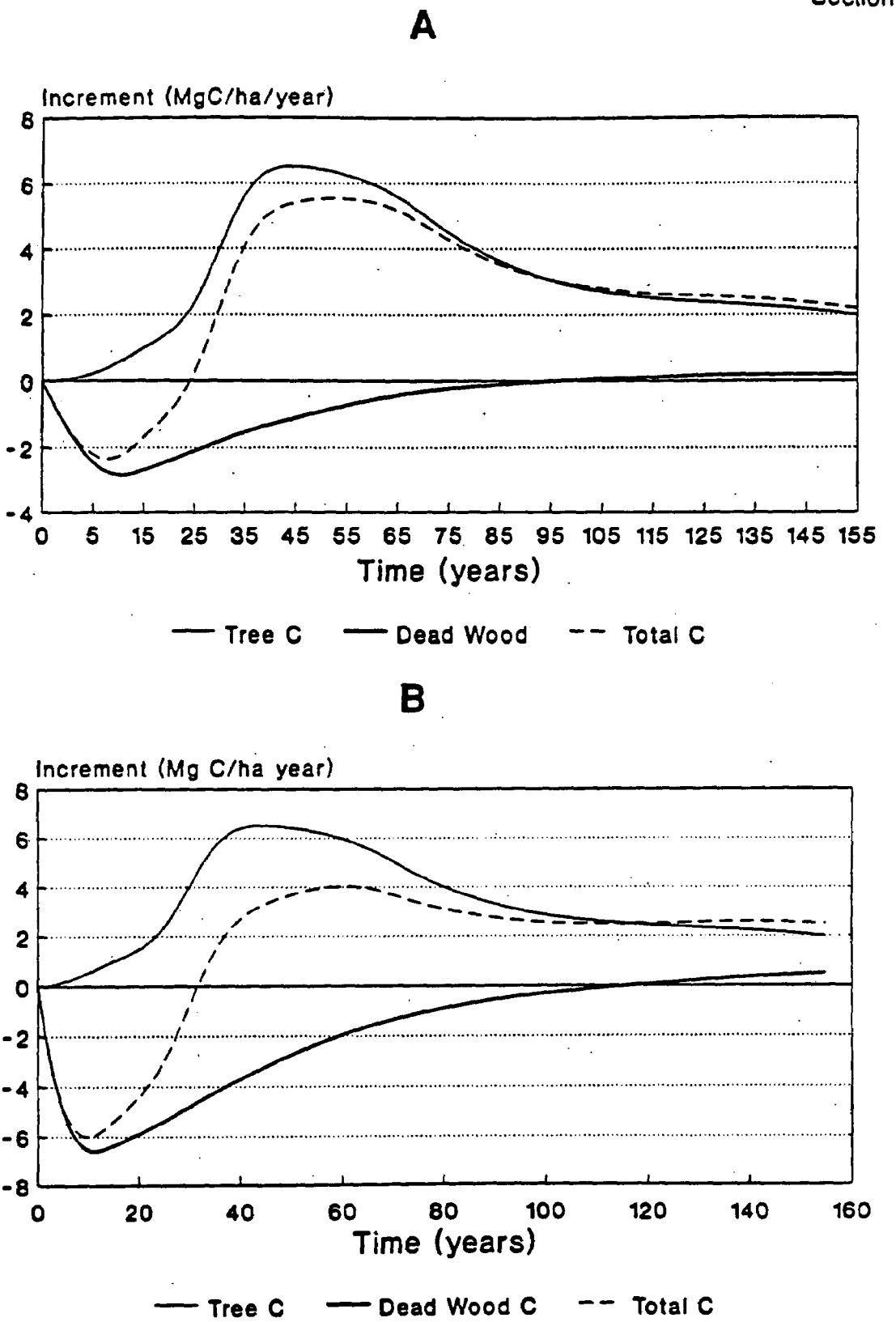

Figure 9.C.2. Carbon increment associated with tree growth and woody detritus for Pacific northwest Douglas-fir forests. A. minimum case and B. maximum case.

Table 9.C.5. Estimated steady-state stores of woody detritus for US forests.

Region Ecosystem	Steady-state stores ($g\text{-}C\ m^{-2}$) minimum	maximum	median	Dead/Live Ratio
Pacific Northwest				
Douglas-fir	5902	20030	11658	0.26
Spruce/fir	2143	6667	4082	0.20
Hemlock/spruce	3785	17031	9083	0.17
Lodgepole pine	439	2049	1129	0.13
Hardwoods	952	3047	1758	0.09
Ponderosa pine	3802	6844	5185	0.32
Redwoods	12544	26341	18293	0.20
Rocky Mountains				
Douglas-fir	897	3312	1881	0.12
Ponderosa pine	938	2732	1753	0.16
Spruce/fir	3673	6733	5050	0.45
Larch	3757	6750	4976	0.20
Lodgepole pine	1142	4397	2692	0.21
South/Southeast				
Oak/hickory	975	1710	1302	0.11
Oak/pine	1318	2207	1699	0.18
Bottom land	828	1131	958	0.08
Natural pine	1527	3925	2326	0.19
Planted pine	2443	3819	3033	0.24
Northeast				
White/red pine	903	3716	2028	0.17
Spruce/fir	1862	5187	3192	0.29
Oak/hickory	947	1775	1315	0.07
Maple/beech/birch	1456	2096	1747	0.12
North Central				
White/red pine	2039	8387	4578	0.17
Spruce/fir	2076	6007	3648	0.59
Maple/beech	590	903	753	0.05
Aspen/birch	2163	3533	2690	0.22
Bottom land	2801	4639	3589	0.15
Oak/hickory	987	3618	2114	0.14

forests, the minimum and the maximum steady-state estimates appear to be far below field measurements for this type. Bingham and Sawyer (1988) found 10,000 g-C m^{-2} of logs; inclusion of snags, roots and branches would indicate that a steady-state value of 24,000 g-C m^{-2} could be expected. A major problem with estimating steady-state stores for this type appears to be an extremely low mortality rate, giving a species longevity of over 10,000 years. Setting the mortality turnover rate to be 0.002 to 0.003 yr^{-1} gives reasonable mass inputs and brings the steady-state value closer to the expected number. Another problem is that the maximum biomass reported by Birdsey (1990) for this type is probably less than half the reported maximum. When this difference is also taken into account the steady-state stores are similar to the estimated stores.

Rocky Mountain Forests. Brown and See (1981) and Sacket (1979) measured 1,150 and 1,800 g-C m^{-2} of logs in ponderosa pine forest of Montana and Arizona, respectively. Including snags, coarse roots, and branches, a value of 1,782 to 2,415 g-C m^{-2} would seem reasonable for the total dead wood stores in ponderosa pine forests. This would indicate that the estimated steady-state stores are somewhat low for this type, although the estimated decay rate constants, mortality turnover and mature forest biomass all seem reasonable for this type. The maximum estimate of 17,956 g-C m^{-2} in spruce-fir is considerably higher than the 3,600 g-C m^{-2} of logs and snags measured by Moir et al. (in review) in a 300-year old spruce-fir forest in Colorado, even if an additional 720 g-C m^{-2} is added to correct for total wood mass. In Montana, Brown and See (1981) found 2,650 g-C m^{-2} in logs for spruce-fir forests, while correcting for snags, roots and branches would indicate a total wood store as high as 5,565 g-C m^{-2} would be reasonable for this type. The mortality turnover rate appears to be too high for this forest type (see above) by a factor of two. Reducing the mortality turnover rate for Rocky Mountain spruce-fir forests to 0.005 to 0.0075 yr^{-1} gives more realistic steady-state stores of 3,673 to 6,733 g-C m^{-2}. The estimated median steady-state mass of 4,976 g-C m^{-2} for larch forests is higher than the adjusted total of 4,200 g-C m^{-2} (Brown and See 1981). However, the overall dynamics of the forest appear reasonable. In lodgepole pine forests, both the minimum and maximum steady-state estimates are too high. Fahey (1983) measured 2,100 in logs and snags; Moir et al. (in review) measured 2,000 g-C m^{-2} in logs and snags; and Brown and See (1981) found 1,600 g-C m^{-2} in logs within undisturbed old lodgepole pine forests. Correcting for the missing dead wood components indicates a range of 2,400 to 3,360 g-C m^{-2} would be reasonable for this forest type. The most likely problem for lodgepole pine forests is an overestimate of the mortality turnover rate, which gives mass inputs larger than observed. Reducing the input rates to values observed in lodgepole pine forests of the Pacific Northwest gives a more reasonable median steady-state estimate of 2,692 g-C m^{-2}.

Southeast/South. Oak/hickory forests have been reported to have 1,080 g-C m^{-2} in logs and snags (Muller and Liu 1991), and between 1,050 g-C m^{-2} (Harmon et al. 1986) and 1,200 g-C m^{-2} (Lang and Forman 1978) in logs. Adjusting to total dead wood stores would indicate a range of 1,300 to 2,500 g-C m^{-2}, which is slightly higher than the estimated range for this forest type of 975 to 1,710 g-C m^{-2}. The dynamics of

this forest type may be correct, however, as field studies included the dead woody mass killed by the chestnut blight. In bottom land hardwoods, Onega and Eickmeier (1991) measured 1,000 g-C m^{-2} in logs and snags. Correcting for dead roots and branches indicates the measured mass of 1,200 g-C m^{-2}, which falls slightly above the estimated minimum and maximum of 828 to 1,131 g-C m^{-2}. In southeastern natural pine forests, a range of 551 to 890 g-C m^{-2} of logs has been measured (Harris et al. 1973, Harmon et al. 1986). Correcting to total dead wood stores indicates a range of 1,323 to 2,136 g-C m^{-2}, which is lower than the estimated range of 1,527 to 3,925 g-Cm^{-2}.

Northeast. In maple/beech/birch forests the estimated maximum and minimum is much lower than the observed range. Tritton and Hornbeck (1982) measured 2,500 g-C m^{-2} in logs and snags, while Gore and Patterson (1986) found 1,700 to 2,000 g-C m^{-2} in logs. Adjusting these to the dead wood total indicates a range of 3,000 to 4,200 g-C m^{-2}, which is almost twice the estimated median steady-state store. The mostly likely problem is a poor underestimate of the input rate since New England Forest Surveys do not report mortality rates. However, increasing these rates gives mass input values that exceed observed values. It is also possible that decay rate constants were overestimated, although again there is little data to indicate if this is the case.

North Central. In oak hickory forests McMillan (1981) measured 800 g-C m^{-2} in logs, while adjusting for other types of dead wood indicates a value of 1,680 g-C m^{-2} (which is bracketed by the estimated range of 493 to 1,809 g-C m^{-2}). The initial steady-state estimates for spruce-fir forest in this region are roughly twice that observed in other regions with this forest type. As the decay rate is probably relatively constant for this type, it is most likely that the mortality turnover for spruce-fir in the North Central region is too high. I therefore recommend using a range of 0.007 to 0.013 yr^{-1}.

MODELING WOODY DETRITUS

The parameters presented above can be used to model the woody detrital dynamics of a wide range of forest types. In this section I will illustrate these dynamics by examining the behavior of two forest types, Douglas-fir in the Pacific Northwest and bottom land hardwoods of the Southeast. These two forest types were selected because they represent the extremes of the parameter estimates (Tables 9.C.2 thru 9.C.5). For each of these two types, I simulated a minimum and maximum case of dead wood storage. For the minimum, I used the minimum mortality turnover rate, steady-state stores, and harvest input, as well as the maximum decay rate. For the maximum case, I used the maximum mortality turnover rate, steady-state stores, and harvest input, as well as the minimum decay rate. Thus, the four simulations represent the extremes of behavior both among forest types and within forest types.

In all cases, except the bottom land hardwood minimum, the addition of dead wood dynamics greatly altered the carbon flux dynamics of harvested forests (Figures 9.C.3 and 9.C.4). For Douglas-fir forests, the addition of woody detritus makes the harvested forest a net source of carbon for approximately 25 years (Figure 9.C.3). Moreover,

these cumulative losses are not balanced by the regrowing trees until the forest is 35 to 60-years old. The addition of woody detritus also influenced the behavior of bottom land hardwood forests, but to a lesser extent than Douglas-fir forests (Figure 9.C.4). Nonetheless, in the case of the bottom land hardwood maximum, addition of dead wood dynamics made the harvested system a net source of carbon for the first 15 years, and these cumulative losses were not offset by new growth until the forest was 27-years old.

In addition to increasing the net source immediately following harvest, the addition of woody detritus increased the net sink in the later stages of succession. These increases ranged from approximately 10% in the bottom land hardwood minimum case to 20% in the Douglas-fir maximum case. While these increases may not seem important, these low rates of increased uptake extend over many decades and may greatly increase the overall sink of older forests.

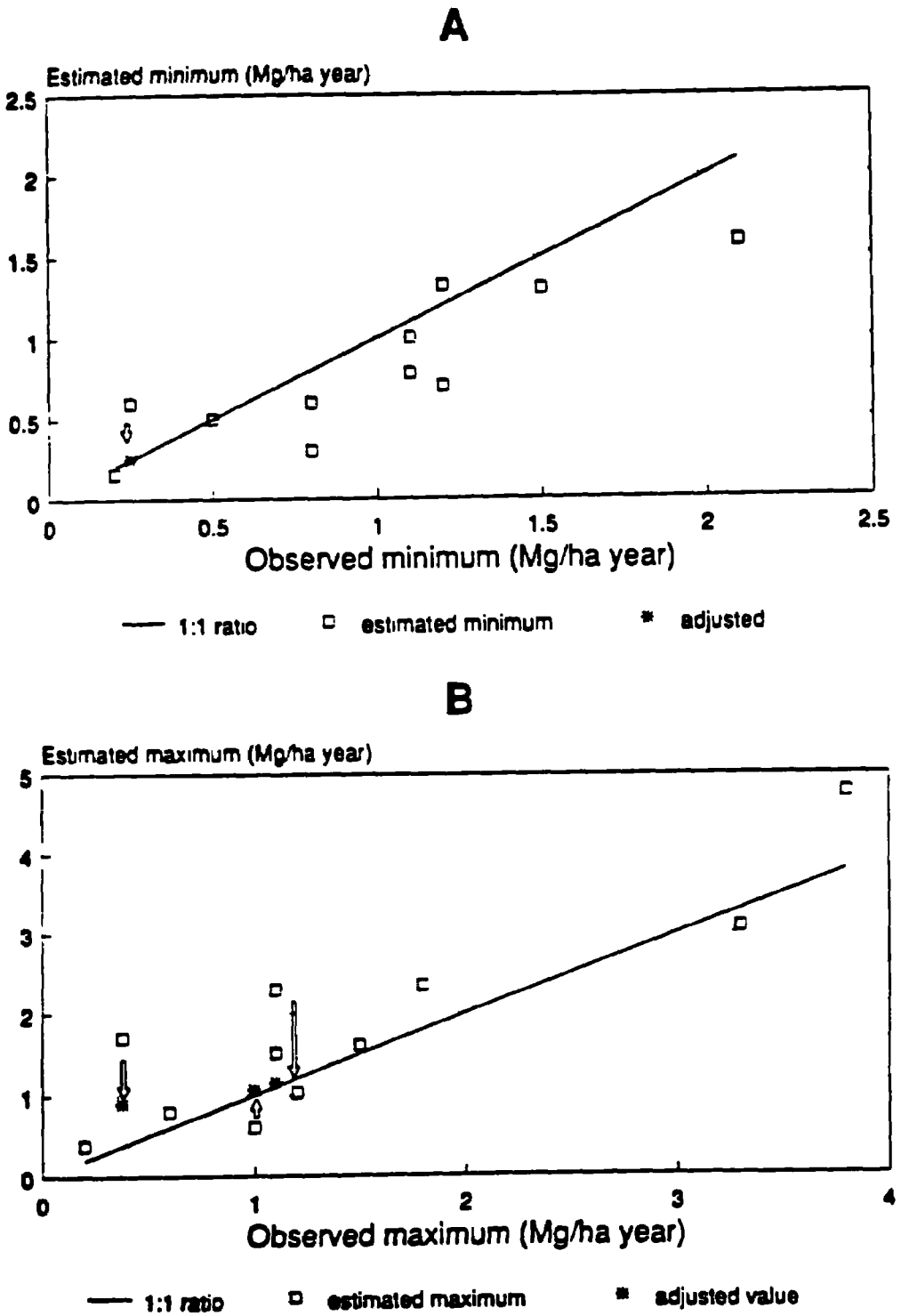

Figure 9.C.3. Observed versus estimated mass inputs of woody detritus for US mature to old-growth forests. The arrows indicate mortality turnover rates that were adjusted A. minimum mass inputs, B. maximum mass inputs.

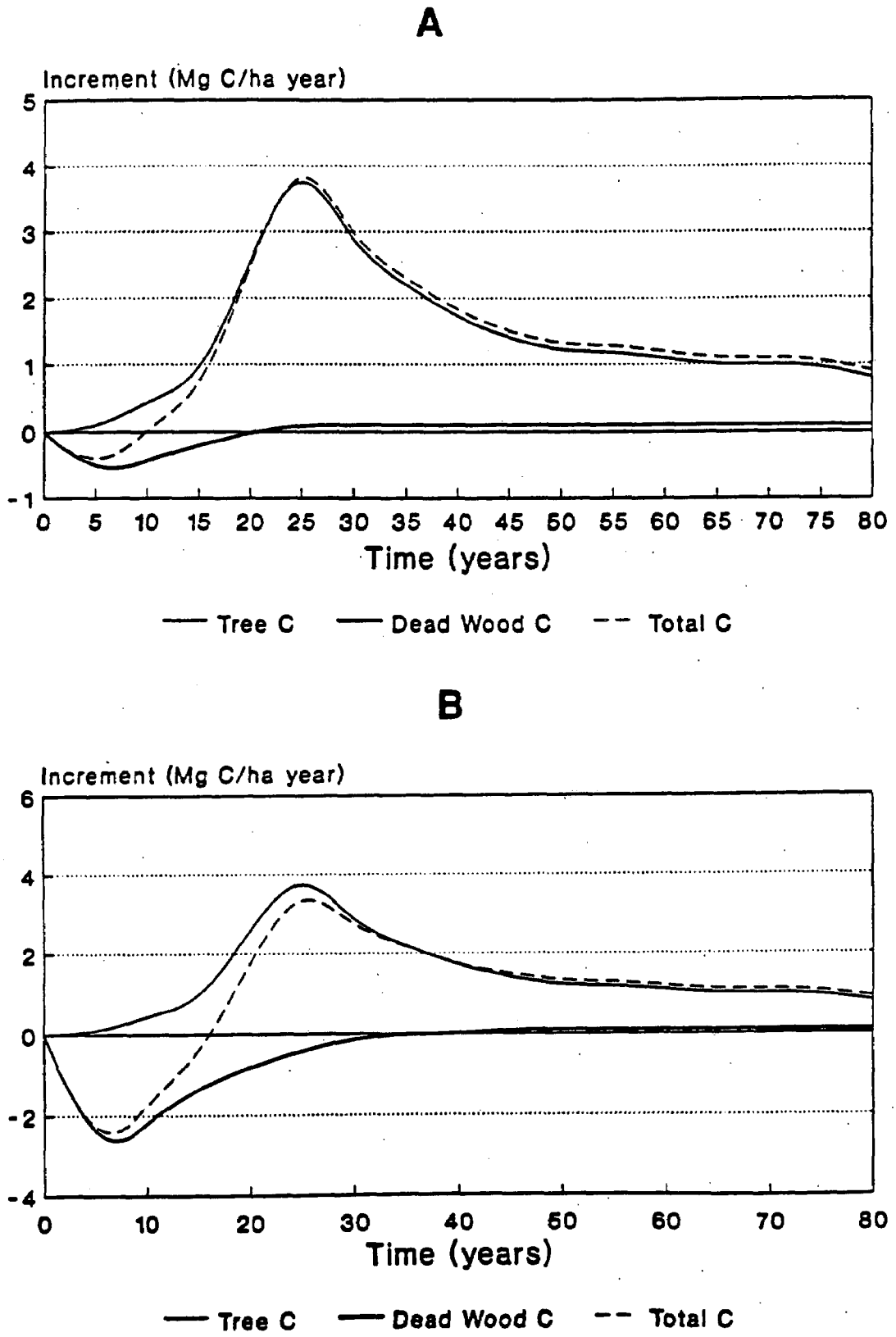

Figure 9.C.4. Carbon increment associated with tree growth and woody detritus for southeastern bottom land hardwood forests. A. minimum case and B. maximum case.

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D. Net Primary Productivity of Non-forested Lands

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Carbon flux on forested lands in the United States is summarized in Section 4 of this report. In this section, net primary productivity (NPP) of nonforested lands in the conterminous US is estimated in order to complete a first approximation of NPP in the US. The treatment of the land base in Section 4 and this section is consistent with the data developed by the USDA for the 1989 Resource Protection Act Assessment (USDA 1989). Estimates of NPP are made for the three major nonforest cover types in the US as identified by the Soil Conservation Service (SCS): rangeland, cropland, and pastureland (USDA, SCS 1987). Estimates of NPP are also made for the remaining nonforested land not included in these three land use types, here termed "other land." This category is described in more detail later in this summary. Data on wetland productivity are also presented, although wetland is not a primary land cover type used by the SCS (USDA, SCS 1987). Wetlands are included primarily under forested lands (for forested wetlands) and in the "all other land" cover type. In addition, some wetlands are also considered to be open water and not described as wetlands by SCS, so the total wetland area presented here is an underestimate of the US total.

Methods

Two general steps were used in estimating nonforest NPP. The areal extent of each land cover type in each state was first estimated, and then the NPP per unit area of those cover types was estimated using averages by state of site specific measurements of NPP. Total annual NPP was calculated by multiplying the area estimates and NPP per unit area estimates together.

All nonfederal land cover estimates are based on the 1982 Natural Resources Inventory (NRI) database (USDA, SCS 1987). The 1982 NRI is a statistically-based sampling of land use on nonfederal lands in the US conducted by the SCS. The sampling strategy was designed to provide natural resource data at a substate level, specifically for Major Land Resource Areas (USDA, SCS 1981). In the analysis presented here, the NRI data are summarized by state, and then aggregated into regions of the US (Table 9.D.1) as defined by the USDA (1989). Since the 1982 NRI did not include federal land, estimates of federal rangeland by region were obtained from the USDA (1989). The area of federally owned pastureland and cropland was assumed to be insignificant.

Different methods were used to estimate the productivity of each land cover type depending on the type and spatial density of the available estimates of site specific NPP, as detailed below. In all cases, dry organic matter weights were divided by two to convert to their carbon equivalent weight.

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Table 9.D.1. Regions of the United States as defined by USDA (1989).

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

*Not included in this study

Rangeland Productivity

Rangeland is defined as "land on which the climax vegetation (potential natural plant community) is predominantly grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing" (USDA SCS 1987). Thus rangelands include the tall, mixed, and short grass prairies of the central US, as well as shrublands in the Great Basin, deserts in the Southwest and many wetlands.

Several steps were taken to estimate rangeland NPP. First, the amount of above-ground production for each NRI rangeland sample point was estimated using the Soils-5 interpretation record number and its corresponding estimate of above-ground NPP (ANPP; USDA, SCS 1983). ANPP estimates for favorable, normal, and unfavorable years are made for each Soils-5 interpretation record by the SCS. Favorable, normal and unfavorable are defined based on total annual precipitation. The wettest 10% of the years in the historical record for a site are termed favorable, while the driest 10% of the years are termed unfavorable (USDA, SCS 1976).

Estimates of ANPP for normal years are used in this analysis. A total of 8,272 Soils-5 production estimates were used in estimating rangeland ANPP. Each ANPP estimate was multiplied by the area represented by the NRI sample point to estimate the total annual ANPP for the rangeland represented by that sample point. Mean ANPP per hectare of each USDA region (USDA 1989) was calculated as an area-weighted average of the ANPP values of all the NRI sample points in the region.

The next step was to estimate total NPP from the estimates of ANPP. Few studies have estimated both ANPP and below-ground net primary productivity (BNPP) for grasslands (Coupland 1979). ANPP and BNPP data at nine central grassland sites were averaged together to estimate the proportion of total plant production that occurs underground. At these sites, BNPP is twice as great as ANPP, so the ANPP estimates for rangelands were tripled to estimate total plant NPP.

NRI data and the corresponding Soils-5 interpretation database are only available for non-federal lands. In order to estimate total NPP for all US rangelands, it was assumed that rangeland productivity was the same for federal and nonfederal lands in each region of the US. Thus, estimates of total production on federal rangeland were made by multiplying the total federal rangeland area in each region by the mean NPP of the nonfederal rangeland in the same region.

Cropland Productivity

Cropland productivity was estimated by converting crop yield into total plant production for each state. Crop yields per unit area were obtained for 1984 from USDA (1986) as compiled by Tom Moser (ManTech Environmental Technology, Corvallis, OR, personal communication). Actual yield data for each crop were used for each of the top ten producing states in the US. The mean crop yield per unit area of the top ten states was used as an estimate of the mean yield in the remaining states. Some of the NRI crop cover types are not crop specific, such as "all other row crops". The average NPP of crops for which yield information was available was used to estimate the NPP of these general cover types.

Crop yield was converted to total plant yield using the conversion factors listed in Sharp et al. (1976) or by using the harvest indices from Donald and Hamblin (1976). The conversion factors in Sharp et al. (1976) account for below-ground as well as above-ground production. They do not account for moisture content of the crop. The total plant production was therefore reduced by the percent moisture content of the crop, as estimated by Sharp et al. (1976) or in the American Society of Agricultural Engineers Standards Handbook (ASAE 1991). If harvest index values were used to estimate total plant production, they were modified to estimate below-ground production. After a mean NPP was estimated for each crop, the total primary production for each state was calculated by multiplying the mean NPP by the crop acreage from the 1982 NRI.

Pastureland Productivity

Pastureland is defined by the SCS to include "land used primarily for production of introduced or native forage plants for livestock grazing. Pastureland may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture" (USDA,

SCS 1987). Pastureland can be intensively managed, including fertilization treatments and weed control.

Pastureland productivity for each state was assumed to be equal to that of haylands in the same state. Hayland yield was obtained from USDA (1986), and converted to total plant yield assuming total NPP is twice the above-ground productivity. No data on below-ground productivity in pasturelands or haylands was found for this study, so doubling above-ground productivity to estimate total NPP is viewed as conservative. The productivity data were corrected for moisture content, assuming a moisture content of 14% (Sharp et al. 1976). The areal estimates of pastureland obtained from the 1982 NRI database were multiplied by these productivity estimates to calculate the total primary production for each state.

Other Land Productivity

This category includes the remainder of land cover types recognized in the 1982 NRI database, including "other cropland, other land in farms, barren land, urban land, and rural transportation." This category constitutes about 9.7% of the surface land area of the US. About 25% of this land is unvegetated and thus has no NPP. NPP per unit area of the remainder of this category was assumed to be equal to the combined mean NPP (per unit area) of rangelands, croplands, and pasturelands (3838 kg ha^{-1} yr^{-1}). Total primary production was estimated by multiplying this mean NPP by the total area of other land covered by vegetation.

Wetland Productivity

The area of wetlands in each state in the conterminous US was estimated using the NRI dataset (USDA, SCS 1987). As previously mentioned, this is not an independent land use category in the NRI database, but is a descriptor of one of the other land cover types. About 52% of the wetland NRI samples points are classified as forests, 23% are in the "all other land" category, and 7% are included in the rangeland cover type. Thus, it is inappropriate to include much of the wetland area in the overall estimate of nonforest productivity. Furthermore, the NRI classifies some wetlands as water, so the NRI underestimates wetland area in the US. Nevertheless, an estimate of the productivity of the sampled wetlands in the US was made in order to ascertain the relative importance of wetlands in the overall carbon balance of the United States.

Few estimates of net primary productivity in wetlands are available, and estimates of below-ground production in forested wetlands are especially rare. Productivity estimates were obtained from the summaries by Brinson et al. (1981) and Bradbury and Grace (1983). Estimates were available from 45 different sites, and these data were first averaged by state and then by region. Relative below-ground productivity was estimated using all the data available for the US where below-ground productivity was measured. This proportion was used to estimate total production at sites where below-ground productivity was not measured. Wetland productivity for a state was estimated either using the average productivity for the state as calculated from the data summaries, or using the regional mean productivity if no state data were available.

Sensitivity analysis of relative below-ground productivity

One of the greatest uncertainties in this data analysis is the relative proportion of total plant productivity contributed by below-ground portions of plants. Below-ground productivity can contribute up to 40 to 90% of total plant primary production and is especially important in grasslands (Stanton 1988). Unfortunately, measurements of below-ground productivity are rare. In this analysis, the few data available on below-ground productivity were used to convert estimates of above-ground productivity into total plant productivity. Because these estimates of below-ground productivity are few and imprecise, the effect of changing the relative proportion of below-ground productivity on NPP in rangelands, pasturelands, and croplands was investigated. The sensitivity analysis consisted of calculating total productivity, assuming the below-ground productivity was twice that of above-ground productivity, equal to above-ground productivity, or half above-ground productivity.

Results

Nonforested land composes about 70% of the land surface in the US (USDA, SCS 1987). The NPP of these lands is estimated as 2.03 Pg-C yr^{-1} or 3703 kg-C ha^{-1} yr^{-1} (Table 9.D.2). Rangeland is the dominant land cover type, accounting for about half of the nonforested land (Table 9.D.2, Figure 9.D.1). Cropland is the second largest land cover type, accounting for about a quarter of nonforested land. However, rangelands have the lowest NPP per unit area, averaging less than half the productivity of croplands (Table 9.D.2). Consequently, croplands account for more of the total nonforest NPP than rangelands (Fig. 9.D.2). Pasturelands are intermediate between croplands and rangelands in their NPP per unit area. Other land accounts for less than 15% of the nonforested land area and productivity.

Land cover and productivity data for pasturelands, rangelands, and croplands has been combined by region (Table 9.D.3, Figs. 9.D.3-4). The Rocky Mountain region contains about one third of the above three land cover types, with the South Central, Great Plains, and North Central regions containing over 10% of these land cover types. Only 2% of these land cover types are found in the Northeast.

In terms of total production by region, the South Central region accounts for the highest total NPP. The Rocky Mountains contributes relatively little NPP, considering that it has the largest total area of rangeland, cropland, and pastureland. Mean total NPP for the Rocky Mountains is about half that of the national average for rangelands, croplands, and pasturelands (Table 9.D.3). NPP per unit area is about the same for the easternmost regions (including the Great Plains), but is substantially less for the western regions. Recall though that these values are for nonforest land use types. Forested areas in the West occur at higher elevations with greater rainfall than the lowland areas where the nonforest cover types are located. Thus, forested NPP can be expected to be greater, increasing the NPP per unit area of the western regions. In the East, the climate of nonforested areas does not differ significantly from forested areas, so including forest NPP in the eastern estimate may not change the NPP per unit area values as much as it would in the West.

Table 9.D.2. Summary table of nonforested land productivity

Land Use Type	Area[2] (x 10^6 ha)	Mean Net Primary Productivity (kg-C ha^{-1} yr^{-1})	Total Net Primary Production (Tg-C)
Rangelands	264.8	2427	642.6
Pasturelands	54.0	4763	257.1
Croplands	154.7	5924	916.3
Other Land[1]	74.9	2867	214.7
US Total	548.4	3703[3]	2030.7

1 Other Land = Cropland other (summer fallow, aquaculture, other cropland not planted, grassland in rotation), other land in farms, barren land, other lands, urban and built up land, rural transportation (SCS 1987).

2 Based on 1982 NRI data (SCS 1987) for non-federal land and USDA (1989) for federal rangeland

3 Calculated by dividing the total nonforest net primary production in the US by the total nonforested area in the US

Table 9.D.3. Estimates of net primary productivity for rangelands, croplands and pasturelands by region.

Region[1]	Area (x 10^6 ha)	Mean Net Primary Productivity (kg-C $ha^{-1}yr^{-1}$)	Total Net Primary Production (Tg-C)
Northeast	11.5	5439	62.5
North Central	68.6	5682	390 0
Southeast	15.2	5415	82.2
South Central	98.1	4840	474.6
Great Plains	66.7	5211	347.6
Rocky Mountains	158.6	1986	315 1
Pacific Northwest	34.3	1979	67.8
Pacific Southwest	20.5	3715	76.1
US Total	473.5	3835[2]	1815.9

1 See Table 9.D.1. for list of states in each region (USDA 1989).

2 Calculated by dividing the total net primary production on rangelands, croplands, and pasturelands by the total land area of these cover types in the US.

Relative Distribution of Nonforest Area by Cover Type

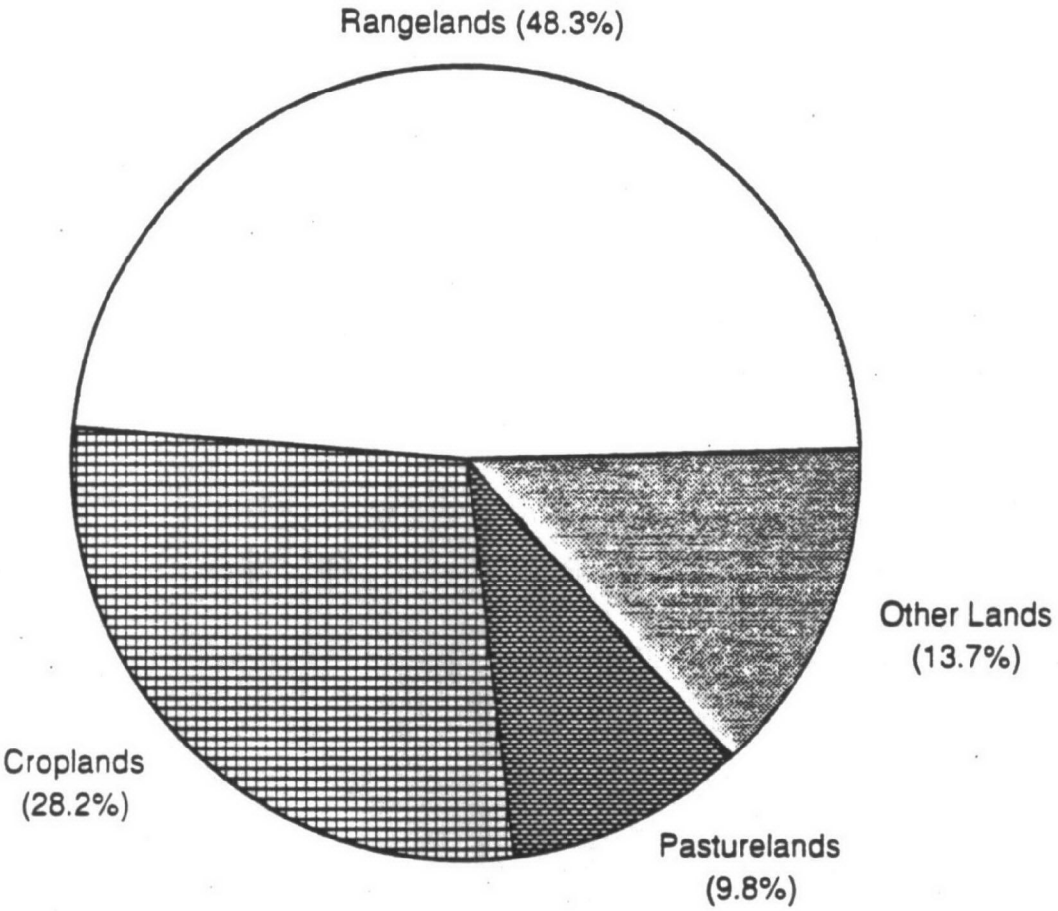

Figure 9.D.1. Relative distribution of nonforested land in the conterminous United States, summarized by cover type. See Table 9.D.2. for absolute estimates of areal coverage of these four principal cover types.

Relative Distribution of Nonforest Net Primary Production by Cover Type

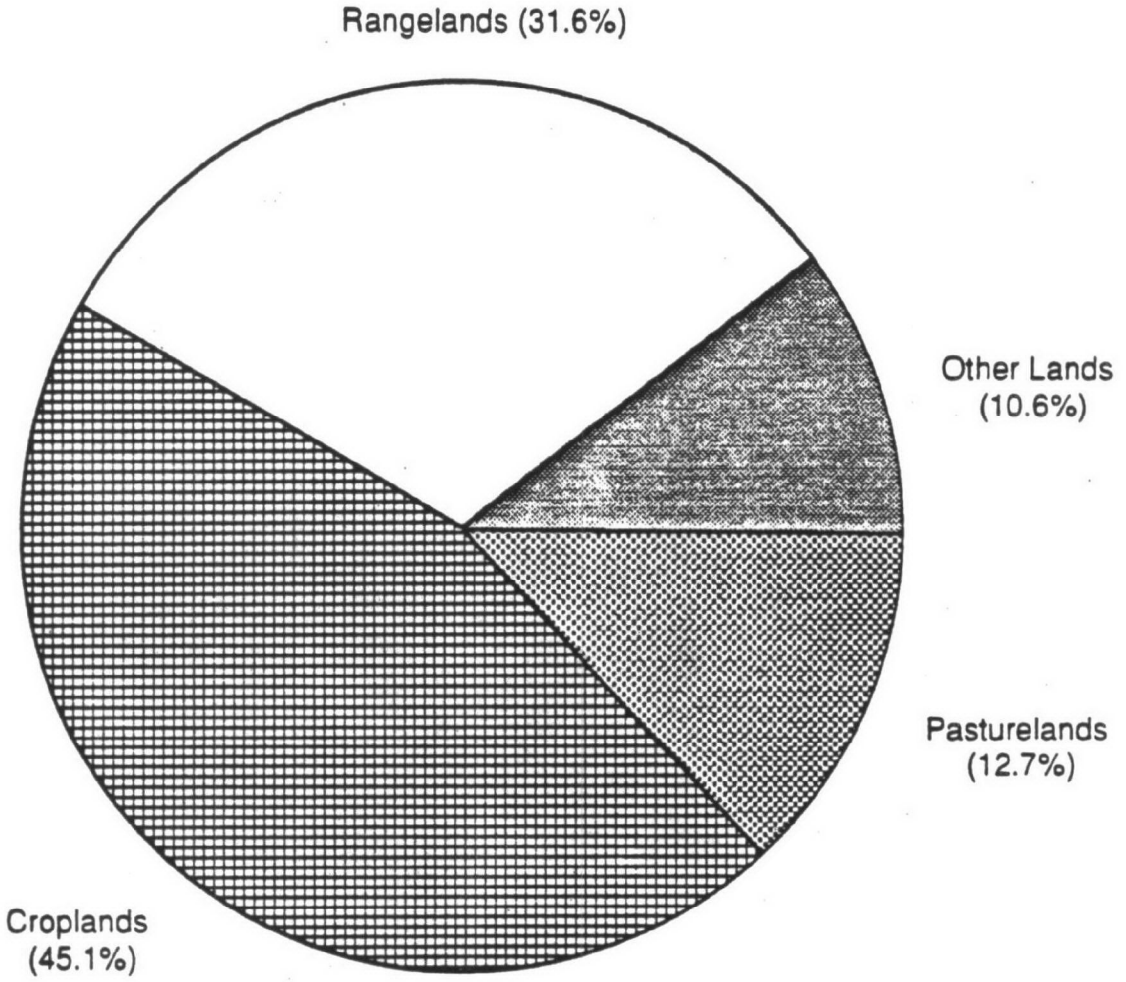

Figure 9.D.2. Relative distribution of nonforest net primary production, summarized by cover type. See Table 9.D.2. for absolute estimates of NPP by cover type.

Relative Distribution of Rangeland, Cropland, and Pastureland Area by Region

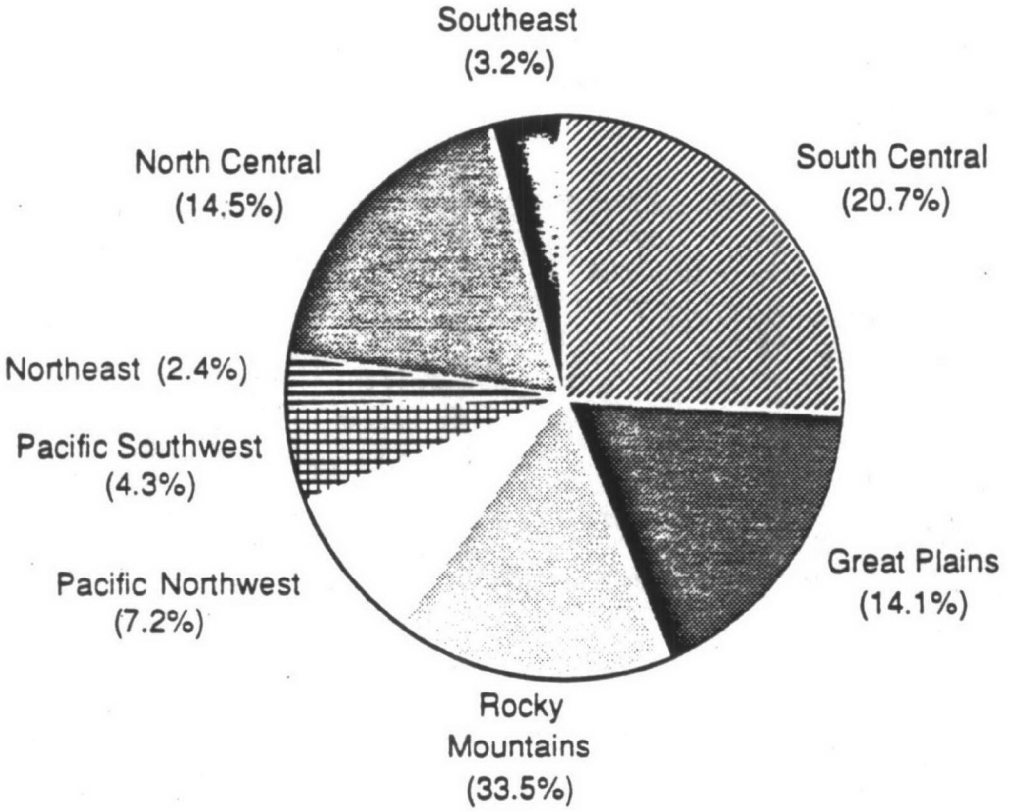

Figure 9.D.3. Relative distribution of rangeland, cropland, and pastureland area in the United States, summarized by region. See Table 9.D.3. for absolute estimates of areal coverage by region.

The NPP data have also been summarized for each land cover type in each region (Tables 9.D.4-6). Over half of the rangeland area occurs in the Rocky Mountains (Table 9.D.4). Most rangeland NPP occurs in the Great Plains, Rocky Mountains, and South Central regions. On a per unit area basis, the highest productivity occurs in the North Central and Southeast regions, where the NPP per unit area is about three times higher than the national average (Table 9.D.4).

About 80% of the cropland in the US occurs in the North Central, South Central, South East, and Great Plains regions (Table 9.D.5). These regions also contribute about 80% of the total crop productivity in the US. NPP per unit area is about the same in all regions, except for the Rocky Mountains being relatively low and the Pacific Southwest (California) being relatively high. The California NPP is high probably because of the intensive use of irrigation.

Almost three quarters of the pastureland in the US occurs in the North Central, Southeast and South Central regions (Table 9.D.6). These three regions account for about the same proportion of total primary production. All regions have about the same NPP per unit area, except for the Pacific Southwest where the estimated NPP is about twice the national average, again, probably because of the intensive use of irrigation.

Wetlands constitute about 4% of the land in the US (Table 9.D.7) as sampled by the 1982 NRI. Total estimated NPP for wetlands is about 11% of the total estimated for nonforested land. Most of the wetland area and production is concentrated in the North Central, South Central, and Southeast regions. Recall, however, that these values are underestimates, because some wetland areas are considered open water in the NRI.

Discussion

Our estimate of nonforest NPP can be combined with information on forest NPP from Section 4 to estimate total NPP for the US. The total annual wood production from the analysis in Section 4.B is 337 Tg-C yr^{-1}. For a very rough approximation that number could be tripled to account for foliage and fine root production. This would bring the total national NPP (forest plus nonforest) estimate to 3.04 Pg-C yr^{-1}.

This estimate can be compared with modeled values of NPP for potential natural vegetation in North America recently completed by McGuire et al. (1992). They estimated a total NPP of 7.03 Pg-C yr^{-1} for all of North America. The conterminous US comprises about 37.8% of North America, and assuming that NPP is uniform across North America, the US should account for 2.66 Pg-C yr^{-1} of the modeled NPP. Our estimate of 2.84 Pg-C yr^{-1} NPP in the US is consistent with this estimate. Furthermore, McGuire et al. (1992) simulated a NPP per unit area of 3,387 kg-C ha^{-1} yr^{-1}. Our estimate of nonforest NPP is 3, 703 kg-C ha^{-1} yr^{-1}, which falls within 10% of the modeled estimate for North America. NPP per unit area is highest in the eastern US in the model simulations, as it is in the estimates made here. Rangeland is the least disturbed nonforest land cover type, and thus is probably most appropriate to compare with equivalent land cover types in the potential natural vegetation classification used

Relative Distribution of Rangeland, Cropland, and Pastureland Net Primary Production by Region

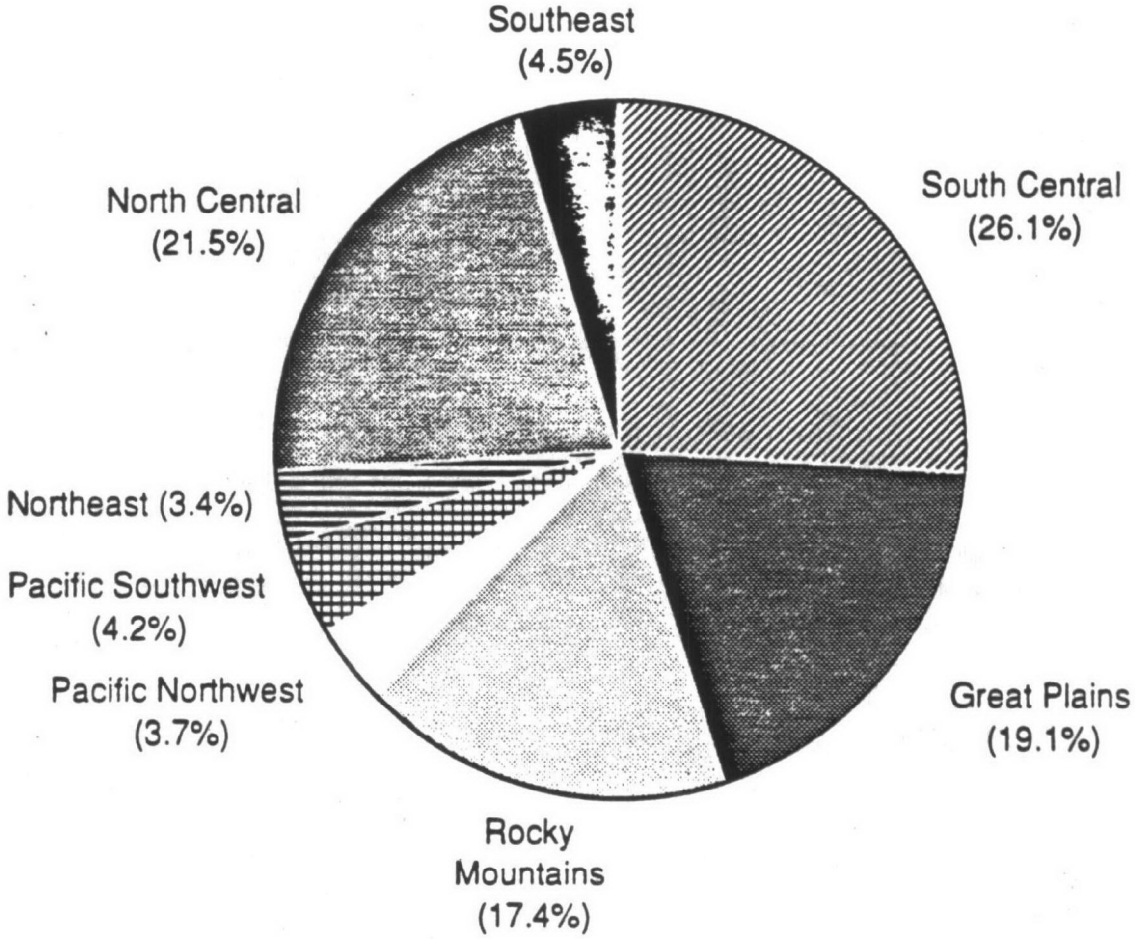

Figure 9.D.4. Relative distribution of net primary production of rangeland, cropland, and pastureland in the United States, summarized by region. See Table 9.D.3. for absolute estimates of NPP by region.

Table 9.D.4. Estimate of net primary productivity for federal and non-federal rangelands in the conterminous US.

Region	Area[1] (x 10^6 ha)	Mean Net Primary Productivity (kg-C ha^{-1} yr^{-1})	Total Net Primary Production[2] (Tg-C)
Northeast	0	0	0
North Central	0.2	7947	1.2
Southeast	0.6	7878	4.8
South Central	45.2	4053	183.1
Great Plains	31.2	4406	137.4
Rocky Mountains	142.5	1663	237.0
Pacific Northwest	29.2	1381	40.3
Pacific Southwest	160.0	2428	38.8
US Total	264.8	2427[3]	642.6

1 Based on 1982 NRI data (USDA, SCS 1987) for non-federal rangelands and USDA (1989) for federal rangelands

2 Based on productivity estimates from Soils-5 (USDA, SCS 1983) interpretation database (SCS 1983).

3 Calculated by dividing the total net primary production on rangelands by total rangeland area in the US.

Table 9.D.5. Estimate of net primary productivity for croplands in the conterminous US.

Region	Area[1] (x 10^6 ha)		Total Net Primary Production[2] (Tg-C)
Northeast	7.1	5816	41.4
North Central	54 1	5862	317.0
Southeast	8.9	5538	49.1
South Central	31.9	6209	197.8
Great Plains	32.1	6105	195.8
Rocky Mountains	13.2	4815	63.0
Pacific Northwest	3.7	5609	20.8
Pacific Southwest	3.9	8043	31.5
US Total	154.7	5924[3]	916.3

1 Based on 1982 NRI data (USDA, SCS 1987) for non-federal land.

2 Based on crop yields form USDA (1986) and conversion factors listed in Sharp et al. (1976) and Donald and Hamblin (1976).

3 Calculated by dividing total net primary production on croplands by total cropland area in the US.

Table 9.D.6. Estimate of net primary productivity for pasturelands in the conterminous United States.

Region	Area[1] x 10^6 ha	Mean Net Primary Productivity[2] (kg-C ha^{-1} yr^{-1})	Total Net Primary Productivity (Tg-C)
Northeast	4.4	4825	21.1
North Central	14.4	4983	71.9
Southeast	5.7	4962	28.4
South Central	21.0	4457	93.7
Great Plains	3.4	4171	14.4
Rocky Mountains	3.1	4962	15.1
Pacific Northwest	1.4	4962	6.7
Pacific Southwest	0.6	9916	5.8
US Total	54.0	4763[3]	257.1

1 Based on 1982 NRI data (USDA, SCS 1987) for non-federal lands.
2 Based on hayland productivity given in USDA (1986)
3 Calculated by dividing total primary production on pasturelands by total pastureland area in the US.

Table 9.D.7. Estimate of net primary productivity for wetlands in the conterminous US.

Region	Area[1] (x 10^6 ha)	Mean Net Primary Productivity (kg-C ha^{-1} yr^{-1})	Total Net Primary Productivity[2] (Tg-C)
Northeast	3.2	8264	26 4
North Central	7.6	8088	61.5
Southeast	8.2	6141	50 4
South Central	7.3	8663	63.2
Great Plains	2.1	2795	5.9
Rocky Mountains	1.4	11949	16.1
Pacific Northwest	0.5	2618	1.3
Pacific Southwest	0.6	2618	1.5
US Total	30.8	7346[3]	226.3

1 Based on 1982 NRI data (USDA, SCS 1987) for non-federal land.
2 Based on productivity estimates from Bronson et al. (1981) and Bradbury and Grace (1983) database
3 Calculated by dividing total net primary production on wetlands by the total wetland area in the US

by McGuire et al. (1992). Our estimate of rangeland NPP falls between the maximum and minimum values of NPP estimated for short and tall grasslands and arid shrublands by McGuire et al. (1992)

The rangeland estimates in the Great Plains can also be compared to those of Sala et al. (1988), who used a similar NPP database based on the SCS data to estimate ANPP (Joyce et al. 1986). The mean ANPP of the Great Plains region calculated here (1,470 kg-C ha^{-1} equivalent to 294 g m^{-2} of biomass) is consistent with the location of the 300 g m^{-2} isopleth in Sala et al. (1988).

That the estimates presented here are comparable to the modeled values is encouraging but not that surprising given that the Terrestrial Ecosystem Model (TEM) model of McGuire et al. (1992) was partially calibrated using actual measured estimates of NPP at specific locations in North America. The more significant question is how well do the measured values of NPP correspond with actual NPP, which is discussed next.

Productivity estimates of grasses, herbs and crops are primarily based on the change in plant biomass during the growing season (e.g., following methods developed by the International Biological Program [IBP]; Cooper 1975, Coupland 1979). Long et al. (1989) demonstrated that this method can underestimate NPP up to fivefold because it does not account for biomass turnover between measurements. This is most critical in regions that are relatively aseasonal (Long and Hutchin 1991). The revised techniques of Long et al. (1989) have not been used to measure NPP in Great Plains grasslands as of yet, so the degree to which the IBP methods underestimate NPP in the US is uncertain.
The relative allocation of NPP above and below-ground is the most uncertain aspect of these analyses because there are relatively few measurements of below-ground production. Changing the relative partitioning of above and below-ground production has a large effect on mean NPP per unit area. For instance, if one assumes that below-ground production is 50% of above ground production, mean NPP in rangelands is 1,220 kg-C ha^{-1} yr^{-1}. However, if one assumes that below-ground production is twice that of above-ground production, than mean NPP becomes 2,430 kg-C ha^{-1} yr^{-1}, double the previous estimate. Data from IBP studies in the prairie suggest that BNPP is twice that of ANPP (Coupland 1979). Consequently, this value was used to convert the rangeland ANPP data to estimates of NPP. Our use of a high proportion of BNPP relative to NPP in grasslands is supported by Stanton (1988). However, recent isotopic measurements suggest that grassland BNPP is more likely equal to ANPP than double ANPP (Milchunas and Lauenroth 1992). Additional measurements and evaluation of methodologies are required to improve BNPP estimates, and thus estimates of total plant NPP.

The estimates of cropland production given here are probably the most precise of all four land types because of the detailed yield data available on a state by state basis. Still, the estimate could be greatly improved by using region-specific conversion factors from crop yield to plant production. Also, better estimates are needed of moisture content of specific crops. The rangeland ANPP data is also relatively precise, because of the large number of ANPP estimates available from the Soils-5 database. Pastureland and other land production estimates are the least precise. No direct

estimates of pastureland NPP were used in this analysis. The other land NPP estimate was made assuming that vegetated land in that category has the same NPP per unit area as nonforest land in the US. For instance, this estimate could be improved by obtaining data on urban land NPP.

The wetland estimate is only a first approximation since the land cover data are incomplete for wetlands. More wetland productivity data are also required.

These nonforest NPP estimates are derived from a mixture of productivity estimates based on average climate conditions (i.e., rangeland data) and from estimates made for a specific year (i.e., cropland data). Sala et al. (1988) has shown that the difference in NPP between favorable and unfavorable years can be 30 to 90% of the average NPP. Consequently, annual NPP can be expected to vary significantly year to year, depending on the differences in seasonal weather conditions. Interannual variability of NPP could be estimated using currently available data, and would lead to a fuller understanding of biospheric carbon dynamics.

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E. Glossary

Age class[1]. The midpoint of an interval of stand age, where stand age is based on trees in the stand sampled at breast height.

Annual mortality[2]: The volume of sound wood in trees that died from natural causes during a specified year.

Annual removals[2] : The net volume of trees removed from the inventory during a specified year by harvesting, cultural operations such as timber stand improvement, or land clearing.

ATLAS: A USDA Forest Service forest inventory model ATLAS allocates prescribed harvest levels among forest types and age classes and updates forest inventory (growing stock volumes) over time At present only lands under private ownership are treated

Base year The reference year for which carbon pools and flux are estimated

Breast height. Point on tree approximately 1 37 m (4-1/2 feet) above ground

Bureau of Land Management (BLM)[2]. An ownership class of Federal lands administered by the Bureau of Land Management, US Department of the Interior.

Coarse material[2] : Wood residues suitable for chipping, such as slabs, edgings, and trimmings.

Commercial species[2] : Tree species suitable for industrial wood products.

Cull tree[2] : A live tree, 12.5 cm (5.0 inches) in diameter at breast height (dbh) or larger, that is unmerchantable for sawlogs now or prospectively because of rot, roughness, or species. (See definitions for rotten and rough trees.)

Diameter class[2] : A classification of trees based on diameter outside bark measured at breast height Dbh is the common abbreviation for "diameter at breast height." With 5 cm (2 inch) diameter classes, the 18 cm (6 inch) class, for example, includes trees 12.5 through 17.2 cm (5.0 through 6.9 inches) dbh, inclusive.

Douglas-fir subregion[2]: The area in Oregon and Washington that is west of the crest of the Cascade Range.

Farmer[2] : An ownership class of private lands owned by a person who operates a farm, either personally doing the work or directly supervising the work.

Federal[2] : An ownership class of public lands owned by the US Government.

Fiber product[2] : Products derived from wood and bark residues, such as pulp, composition board products, and wood chips for export

Fine materials[2] : Wood residues not suitable for chipping, such as planer shavings and sawdust

Forest floor. All dead organic matter above the mineral soil horizon except woody debris.

Forest Industry[2] : An ownership class of private lands owned by companies or individuals operating wood-using plants

Forest Inventory and Analysis (FIA). USDA Forest Service Inventory work group

Forest land[2]: Land at least 10% (16 7% in southeast Alaska) stocked by forest trees of any size, including land that formerly had such tree cover, and that will be naturally or artificially regenerated. Forest land includes transition zones, such as areas between heavily forested and nonforested lands that are at least 10% stocked with forest trees, and forest areas adjacent to urban and built-up lands. Also included are pinyon-juniper and chaparral areas in the West and afforested areas. The minimum area for classification of forest land is one acre. Roadside, streamside, and shelterbelt strips of timber must have a crown width at least 36 m (120 feet) to qualify as forest land. Unimproved roads and trails, streams, and clearings in forest areas are classified as forest if less than 36 m (120 feet) wide.

Forest type[2]: A classification of forest land based on the species presently forming a plurality of the live-tree stocking.

Fuelwood[2]: Wood used for conversion to some form of energy, primarily residential use.

Growing stock[2]: A classification of timber inventory that includes live trees of commercial species meeting specified standards of quality or vigor. Cull trees are excluded When associated with volume, includes only trees 12.5 cm (5.0 inches) dbh and larger.

Hardwood[2]: A dicotyledonous tree, usually broad-leafed and deciduous.

Industrial wood[2]: All commercial roundwood products except fuelwood.

Land area[2] (a) Bureau of the Census: The area of dry land and land temporarily or partly covered by water, such as marshes, swamps, and river food plains; streams, sloughs, estuaries, and canals less than 200 m (one-eighth statute mile) wide; and lakes, reservoirs, and ponds less than 40 acres (16 ha) in area (b) Forest Inventory and Analysis: Same as (a) except that the minimum width of streams, etc., is 36 m (120 feet), and the minimum size of lakes, etc., is 0 4 ha (1 acre). This latter definition is the one used in this publication

Litter: Recently fallen and decomposing organic matter above the mineral soil Does not include woody material greater than 2 cm diameter.

Logging residues[2]: Downed and dead wood volume left on the ground after trees have been cut on timberland.

Merchantable: Trees of diameter greater than 12.5 cm except cull trees.

Merchantable bole volume. Merchantable wood volume in the boles. Includes commercial and noncommercial species.

Merchantable bole carbon: Tree merchantable bole volume expressed as carbon. The proportions of hardwood and softwood and the relevant specific gravity and percent carbon are used to make the conversion.

National Forest (NF)[2]: An ownership class of Federal lands, designated by Executive Order or statute as National Forests or purchase units, and other lands under the administration of the Forest Service including experimental areas and Bankhead-Jones Title III lands.

Net annual growth[2]. The net increase in the volume of trees during a specified year. Components include the increment in net volume of trees at the beginning of the specific year surviving to its end, plus the net volume of trees reaching the minimum size class during the year, minus the volume of trees that died during the year, and minus the net volume of trees that became cull trees during the year.

Net cubic volume[2]: The gross volume in meters (cubic feet) less deductions for rot, roughness, and poor form. Volume is computed for the central stem from a 30 cm (1 foot) stump to a minimum 10 cm (4.0 inch) top diameter outside bark, or to the point where the central stem breaks into limbs.

Non-Federal (Private): Lands owned by private individuals (e.g., farmers), private industries, and Native and other corporations

Non commercial species[2]: Tree species of typically small size, poor form, or inferior quality, which normally do not develop into trees suitable for industrial wood products.

Nonforest land[2]: Land that has never supported forests and lands formerly forested where use of timber management is precluded by development for other uses (Note: Includes area used for crops, improved pasture, residential areas, city parks, improved roads of any width and adjoining clearings, powerline clearings of any width, and 0.4 to 16 ha (1- to 40-acre) areas of water classified by the Bureau of the Census as land. If intermingled in forest areas, unimproved roads and nonforest strips must be more than 36 m (120 feet) wide, and clearings, etc., more than 0.4 ha (1 acre), to qualify as nonforest land.)

Nonstocked areas[2]: Timberland less than 10% (16.7% in southeast Alaska) stocked with growing-stock trees.

Old Growth stand: A stand originating from natural disturbance, with a stand age greater than those specified in the RPA yield tables.

Other Federal[2]: An ownership class of

Federal lands other than those administered by the Forest Service or the Bureau of Land Management.

Other forest land[2]: Forest land other than timberland and reserved timberland. It includes available and reserved unproductive forest land, which is incapable of producing annually 1.4 cubic meters per ha per year (20 cubic feet per acre) of industrial wood under natural conditions because of adverse site conditions such as sterile soils, dry climate, poor drainage, high elevation, steepness, or rockiness.

Other land[2]: Nonforest land less the area in streams, sloughs, estuaries, and canals between 36 and 198 m (120 and 660 feet) wide and lakes, reservoirs, and ponds between 0.4 and 16 ha (1 and 40 acres) in area (that is, nonforest land less non-Census water area)

Other private[2]: An ownership class of private lands that are not owned by forest industry or farmers.

Other products[2]: A miscellaneous category of roundwood products that includes such items as cooperage, pilings, poles, posts, shakes, shingles, board mills, charcoal, and export logs.

Other public[2]: An ownership class that includes all public lands except National Forest.

Other red oaks[2]: A group of species in the genus Quercus that includes scarlet oak, northern-pin oak, southern red oak, bear oak, shingle oak, laurel oak, blackjack oak, water oak, pin oak, willow oak, and black oak.

Other removals[2]: Unutilized wood volume from cut or otherwise killed growing stock, from non-growing stock sources on timberland (for example, precommercial thinnings), or from timberland clearing. Does not include volume removed from inventory through reclassification of timberland to reserved timberland.

Other sources[2]: Sources of roundwood products that are non- growing stock. These include salvable dead trees, rough and rotten trees, trees of noncommercial species, trees less than 12.5 cm (5.0 inches) dbh, tops, and roundwood harvested from nonforest land (e.g., fence rows).

Other white oaks[2]. A group of species in the genus Quercus that includes overcup oak, chestnut oak, and post oak.

Ovendry ton: One dry ton, 907 kg (2000 pounds) of biomass.

Ownership[2]: The property owned by one ownership unit, including all parcels of land in the United States.

Ownership unit[2]: A classification of ownership encompassing all types of legal entities having an ownership interest in land, regardless of the number of people involved. A unit may be an individual; a combination of persons; a legal entity such as a corporation, partnership, club, or trust; or a public agency. An ownership unit has control of a parcel or group of parcels of land.

Pg: Petagram = 10^{15} g = billion metric tons = gigaton

Ponderosa pine subregion[2]: The area in Oregon and Washington that is east of the crest of the Cascade Range.

Private. An ownership class which includes forest industry, farmers, and other private lands

Productivity class[2]: A classification of forest land in terms of potential annual cubic meters per ha (cubic-foot volume growth per acre) at culmination of mean annual increment in fully stocked natural stands.

Public: An ownership class which includes National Forest and other public lands.

Pulpwood[2]: Roundwood, whole-tree chips, or wood residues that are used for the production of wood pulp.

Reserved timberland[2]: Forest land that would otherwise be classified as timberland except that it is withdrawn from timber utilization by statute or administrative regulation.

Roots: Tree carbon below the ground level

Residues[2]: Bark and woody materials that are generated in primary wood-using mills when roundwood products are converted to other products. Examples are slabs, edgings, trimmings, miscuts, sawdust, shavings, veneer cores and clippings, and pulp screenings.

Includes bark residues and wood residues (both coarse and fine materials) but excludes logging residues

Rotten tree[2]. A live tree of commercial species that does not contain a sawlog now or prospectively primarily because of rot (that is, when rot accounts for more than 50% of the total cull volume)

Roundwood products[2]: Logs, bolts, and other round timber generated from harvesting trees for industrial or consumer use.

Saplings Live trees between 0 4 cm and 2.3 cm in diameter at breast height

Sawlog[2] A log meeting minimum standards of diameter, length, and defect, including logs at least 2 4 m (8 feet) long sound and straight, and with a minimum diameter inside bark of 15 cm (6 inches) for softwoods and 20 cm (8 inches) for hardwoods, or meeting other combinations of size and defect specified by regional standards

Sawtimber[2]: A classification of timber inventory that is composed of sawtimber trees

Sawtimber trees[2]. Live trees of commercial species containing at least one 3.6 m (12 foot) sawlog or two noncontiguous 2 4 m (8 foot) logs, and meeting regional specifications for freedom from defect Softwood trees must be at least 22 5 cm (9 0 inches) [27.5 cm (11.0 inches) in Alaska] dbh, and hardwood trees must be at least 27.5 cm (11.0 inches) dbh

Second Growth stands Originating from harvest of old growth stands

Softwood[2]: A coniferous tree, usually evergreen, having needles or scalelike leaves.

Soil organic carbon: Absolute weight of carbon to a one meter depth. Includes all carbon in the less than 2 mm fraction organic.

State[2]: An ownership class of public lands owned by states or lands leased by states for more than 50 years.

Stocking[2]: The degree of occupancy of land by trees, measured by basal area or number of trees by size and spacing, or both, compared to a stocking standard; that is, the basal area or number of trees, or both, required to fully utilize the growth potential of the land.

Stumps Woody biomass between the roots and the level of the cut.

Tg Teragram = 10^{12} g = million metric tons = megaton

Third Growth stands: Originating from harvest of second growth stands.

Timberland[2]. Forest land that is producing or is capable of producing, crops of industrial wood and not withdrawn from timber utilization by statute or administrative regulation (Note Areas qualifying as timberland are capable of producing in excess of 1.4 cubic meters per ha (20 cubic feet per acre) per year of industrial wood in natural stands. Currently inaccessible and inoperable areas are included)

Tops[2]: The wood of a tree above the merchantable height (or above the point on the stem 10 cm (4.0 inches) diameter outside bark It includes the usable material in the uppermost stem.

Total biomass carbon: Total tree carbon plus sapling carbon.

Total tree carbon: All living tree carbon including roots, tops, and branches and cull trees Does not include saplings

Tree carbon. All living tree carbon including roots, tops, and branches and cull trees.

Unreserved forest land[2]: Forest land that is not withdrawn from use by statute or administrative regulation. Also termed available.

Understory: All live vegetation except that defined as merchantable tree carbon and saplings.

Veneer log[2]: A roundwood product from which veneer is sliced or sawn and that usually meets certain standards of minimum diameter and length and maximum defect.

Weight[2]: The weight of wood and bark, oven-dry basis (approximately 12% moisture content).

Woodlands Forest land where timber species make less than 10% stocking or forest land that

is incapable of producing crops of industrial wood under natural conditions, because of adverse site conditions.

Woody debris. All dead organic matter above the mineral soil with a diameter greater than two centimeters includes standing dead.

Yield table[2]: A tabular presentation of volume per unit area and other stand characteristics of even-aged stands by age classes, site classes, species, and density.

Major Eastern forest-type groups[3]

White-red-jack pine: Forests in which eastern white pine, red pine, or jack pine, singly or in combination, comprise a plurality of the stocking. Common associates include hemlock, aspen, birch, and maple.

Spruce-fir: Forests in which spruce or true firs, singly or in combination, comprise a plurality of the stocking Common associates include white cedar, tamarack, maple, birch, and hemlock.

Longleaf-slash pine: Forests in which longleaf or slash pine, singly or in combination, comprise a plurality of the stocking Common associates include other southern pines, oak, and gum.

Loblolly-shortleaf pine: Forests in which loblolly pine, shortleaf pine, or southern yellow pines, except longleaf or slash pine, singly or in combination, comprise a plurality of the stocking Common associates include oak, hickory, and gum.

Oak-pine. Forests in which hardwoods (usually upland oaks) comprise a plurality of the stocking, but in which pine or eastern redcedar comprises 25 to 50% of the stocking Common associates include gum, hickory, and yellow-poplar.

Oak-hickory: Forests in which upland oaks or hickory, singly or in combination, comprise a plurality of the stocking except where pines comprise 25 to 50%, in which case the stand is classified as oak-pine. Common associates include yellow-poplar, elm, maple, and black walnut.

Oak-gum-cypress: Bottomland forests in which tupelo, blackgum, sweetgum, oaks, or southern cypress, singly or in combination, comprise a plurality of the stocking except where pines comprise 25 to 50%, in which case the stand is classified as oak-pine. Common associates include cottonwood, willow, ash, elm, hackberry, and maple.

Elm-ash-cottonwood: Forests in which elm, ash, or cottonwood, singly or in combination, comprise a plurality of the stocking. Common associates include willow, sycamore, beech, and maple.

Maple-beech-birch: Forests in which maple, beech, or yellow birch, singly or in combination, comprise a plurality of the stocking. Common associates include hemlock, elm, basswood, and white pine.

Aspen-birch: Forests in which aspen, balsam poplar, paper birch, or gray birch, singly or in combination, comprise a plurality of the stocking Common associates include maple and balsam fir.

Major Western forest-type groups[3]

Douglas-fir: Forests in which Douglas-fir comprises a plurality of the stocking Common associates include western hemlock, western redcedar, the true firs, redwood, ponderosa pine, and larch.

Hemlock-Sitka spruce: Forests in which western hemlock and/or Sitka spruce comprise a plurality of the stocking Common associates include Douglas-fir, silver fir, and western redcedar.

Redwood: Forests in which redwood comprises a plurality of the stocking Common associates include Douglas-fir, grand fir, and tan oak.

Ponderosa pine: Forests in which ponderosa pine comprises a plurality of the stocking. Common associates include Jeffrey pine, sugar pine, limber pine, Arizona pine, Apache pine, Chihuahua pine, Douglas-fir, incense-cedar, and white fir.

Western white pine: Forests in which western white pine comprises a plurality of the stocking. Common associates include western redcedar, larch, white fir, Douglas-fir, lodgepole pine, and Engelmann spruce.

Lodgepole pine: Forests in which lodgepole pine comprises a plurality of the stocking

Common associates include alpine fir, western white pine, Engelmann spruce, aspen, and larch.

Larch: Forests in which western larch comprises a plurality of the stocking Common associates include Douglas-fir, grand fir, western redcedar, and western white pine.

Fir-spruce: Forests in which true firs, Engelmann spruce, or Colorado blue spruce, singly or in combination, comprise a plurality of the stocking Common associates include mountain hemlock and lodgepole pine.

Western hardwoods: Forests in which aspen, red alder, or other western hardwoods, singly or in combination, comprise a plurality of the stocking.

Chaparral: Forests of heavily branched, dwarfed trees or shrubs, usually evergreen, the crown canopy of which at maturity covers more than 50% of the ground and whose primary value is watershed protection. The more common chaparral constituents are species of Quercus, Cercocarpus, Garrya, Ceanothus, Arctostaphylos, and Adenostoma. Types dominated by such shrubs as Artemisia, Chrysothamnus, Purshia, Gutierrezia, or semidesert species are not commonly considered chaparral.

Pinyon-juniper: Forests in which pinyon-pine, or juniper, or both, comprise a plurality of the stocking.

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[1]Husch, B., C. I. Miller and T. W. Beers. 1982. Forest Mensuration, Third Edition, John Wiley & Sons. 402 pp

[2]Extracted, with metric inserts, from. Waddell, K L., D D. Oswald and D. S. Powell. 1989. Forest Statistics of the United States USDA Resource Bulletin PNW-RB-168 Pacific Northwest Research Station, Portland, Oregon

[3]As Defined in Waddell, K. L., D D. Oswald, and D. S. Powell 1989 Forest Statistics of the United States USDA Resource Bulletin PNW-RB-168 Pacific Northwest Research Station, Portland, Oregon.

F. Metric and English Equivalents

Symbol	Metric unit	=	English unit	Symbol
	Length			
cm	1 centimeter	=	0.3937 inch	in
m	1 meter	=	3.28 feet	ft
m	1609 meter	=	1 mile	mi
	Area			
cm^2	1 $centimeter^2$	=	0.155 $inch^2$	in^2
m^2	1 $meter^2$	=	10.764 $feet^2$	ft^2
km^2	1 $kilometer^2$	=	0.386 $mile^2$	mi^2
ha	1 hectare	=	2.471 acre	ac
	Mass (weight)			
g	1 gram	=	0.0353 ounce	oz
kg	1 kilogram	=	2.2046 pound	lb
t	1 metric ton	=	2204.6 pound	lb
	Volume			
m^3	1 $meter^3$	=	35.31 $feet^3$	ft^3
	Examples of Mass or Volume per unit area			
	1 m^3 ha^{-1}	=	14.3 ft^3 ac^{-1}	
	1 m^2 ha^{-1}	=	4.35 ft^2 ac^{-1}	

Taken from: Husch, B., C. I. Miller, and T. W. Beers. 1982. *Forest Mensuration*, 3rd edn. John Wiley & Sons. 402 p.

G. Acronyms

API	American Paper Institute
ATLAS	Aggregate Timberland Assessment System
BGC	Biogeochemistry model
CENTURY	Soil carbon dynamics model
DBH	Diameter at Breast Height
DoC	US Department of Commerce
EPA	Environmental Protection Agency
ERL-C	Environmental Research Laboratory - Corvallis
EROS	Earth Resource Observation System, USGS
FCM	Forest Carbon Model
FIA	Forest Inventory and Analysis, USDA Forest Service
FORCARB	FORest CARBon model
FORENA	Population-based Forest Dynamics model
FORPLAN	FORest PLANning model
GCM	Global Circulation Model
GISS	Goddard Institute for Space Studies
GNP	Gross National Product
IPCC	Intergovernmental Panel on Climate Change
LAI	Leaf Area Index
MAPSS	Mapped Atmosphere Plant Soil System
NPP	Net Primary Production or Net Primary Productivity
NRI	National Resource Inventory
OSB	Oriented Strand Board
OSU	Oregon State University
PET	Potential Evapotranspiration
RPA	Forest and Rangeland Renewable Resource Planning Act of 1974
TAMM	Timber Assessment Market Model
TREGRO	Physiologically-based Tree Growth Model
UKMO	United Kingdom Meteorological Office
US	United States
USDA	United States Department of Agriculture
USGS	United States Geological Survey
UNCED	United Nations Conference on Environment and Development