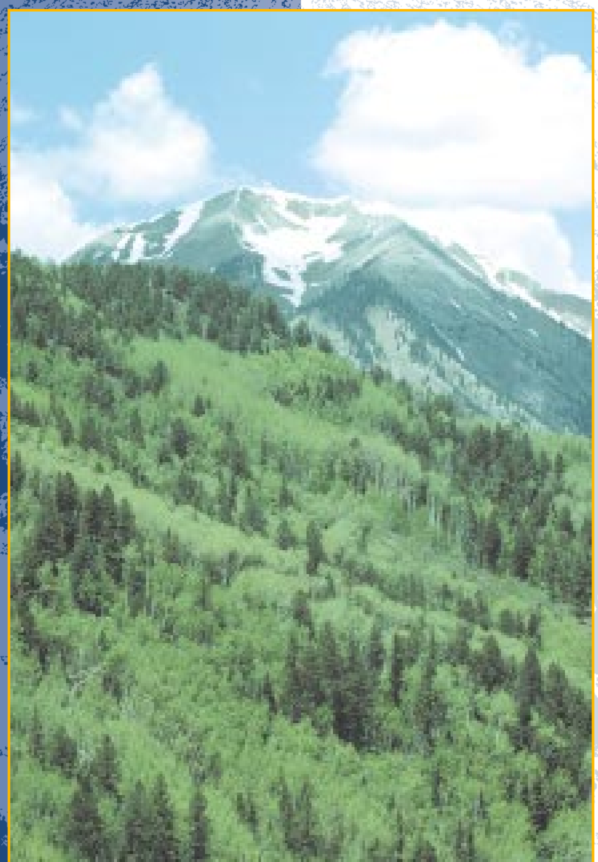
The Impact of Climate Change on America's Forests



A Technical Document Supporting the 2000 USDA Forest Service RPA Assessment

U.S. DEPARTMENT OF AGRICULTURE FOREST SERVICE

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Keywords: forest productivity, vegetation change, carbon sequestration, mitigation, forest sector, timber inventory, soil carbon, carbon accounting, afforestation, deforestation

Abstract

This report documents trends and impacts of climate change on America's forests as required by the Renewable Resources Planning Act of 1974. Recent research on the impact of climate and elevated atmospheric carbon dioxide on plant productivity is synthesized. Modeling analyses explore the potential impact of climate changes on forests, wood products, and carbon in the United States.

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The Impact of Climate Change on America's Forests: A Technical Document Supporting the 2000 USDA Forest Service RPA Assessment

Linda A. Joyce and Richard Birdsey, Technical Editors

Contents

Executive Summary

The increasing concentration of atmospheric carbon dioxide has raised concerns about the vulnerability of forests to potential changes in climate and climate variability. These concerns have prompted governments around the world to commission technical assessments on the impact of climate change on the environment and the economy. Based on the current scientific information within these assessments, governments have initiated negotiations on policy actions to reduce greenhouse gas emissions and to address the vulnerabilities of the ecological, economic, and social systems to climate change. Critical to policy formulation is a periodic synthesis of the ever-expanding knowledge on forest ecology, the impact of climate on forests and of forests on climate, forest management, the socio-economic value of trees and forests, and the role of forests in the global carbon cycle.

The Forest Service conducts periodic assessments of the condition of forest and rangeland resources under the authority of the Renewable Resources Planning Act (RPA). The structure of these periodic assessments allows for the synthesis and integration of the current state of scientific knowledge. As part of the RPA process, this report synthesizes current information that assesses the impact of climate change on U.S. forests. Six policy questions critical to understanding the impact of global climate change on current and future trends[1] form the basis for this report. The first chapter describes mandates and structures of synthesizing scientific information on the forest sector, describes current understandings of the global climate, and closes with the policy questions addressed in this assessment. The next chapters address the six policy questions, which we summarize here.

What are the Likely Effects of Increasing Atmospheric Carbon Dioxide and Prospective Climate Changes on Ecosystem Productivity, as Measured by Changes in Net Primary Productivity?

Joyce and Nungesser (this volume) summarize recent experimental data and modeling analyses that enhance our understanding of the potential impact of carbon dioxide and climate on forests. Analyses of the impact of climate change on forest productivity, based on experimental research and modeling, would suggest that forest productivity may increase under elevated carbon dioxide, but that the local conditions of moisture stress and nutrient availability will strongly temper any response. New field experiments on vegetation and recent meta-analysis studies of the accumulated experimental data demonstrate that elevated carbon dioxide has been shown to increase plant growth by 25-50 percent, with experimental results varying widely. This optimal response is an assumption within the ecological models used to examine the impact of climate change on vegetation. However, when the models are implemented across the geographically diverse landscapes of the United States, the model responses to elevated carbon dioxide are substantially lower, less than 20 percent by ecosystem type reflecting local conditions of moisture stress and nutrient availability. Joyce and Nungesser (this volume) compare the magnitude of these modeled responses to forest growth responses from timber management. They conclude that the growth response to timber management is larger (25 to 39 percent) than the projected productivity response from changes in carbon dioxide or climate (8 to 29 percent).

Joyce and Nungesser (this volume) also examined the implications of two modeling assumptions in the last RPA assessment: 1) climate and vegetation were in equilibrium, and 2) the spatial scale of the ecological model was an adequate scale for national level analyses. Most modeling analyses of the impact of climate change, including the RPA analysis, use input data gridded at a 50-km scale to describe the dominant climate and land surface features of large regions, such as the United States. Joyce and Nungesser concluded that moving the climate change analyses to a finer spatial scale, for example 10-km × 10-km grid size, did not necessarily improve the estimates of the impact of climate change on forest productivity. Analyses done at the 50 km scale were adequate to address national and regional issues, except in areas where a mosaic of forest types occurred or where temperature and precipitation gradients were large. Incorporating land use changes appears to be a critical next step in the analysis of the impact of climate change on forest productivity.

To What Geographic Extent Will Potential Ecosystem Types Change or Move Across the United States, as Measured in Composition and Boundary Changes?

Bachelet and Neilson (this volume) conclude that the following changes could occur in the vegetation distribution across the United States as a result of climate change: 1) boreal forests and taiga-tundra regions are predicted to move northward at the expense of the tundra; 2) warmer

[1] Joyce, L.A.; Birdsey, R.; Mills, J.; Heath, L. 1997. Progress toward an integrated model of the effects of global change on United States forests. In: Birdsey, R.; Mickler, R.; Sandberg, D. [et al.], eds. 1997. USDA Forest Service Global Change Research Program Highlights 1991-1995. Gen. Tech. Rep. NE-237. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 93-96.

scenarios produce the largest impacts on the boreal forest, but may also be responsible for forest dieback on the conterminous U.S.; 3) northwest and southeast forests might initially expand, then later contract in area; and 4) southwestern desert species may move into the Great Basin region. The choice of climate change scenario and the treatment of carbon dioxide effects in each model strongly influence the simulations and the uncertainty in them. Results may indicate the direction of possible change but should not be taken as solid predictions. Disturbance regimes will be affected by climate change, but they are difficult to simulate and affect the outcome of the models. Moreover, important factors not included in these models such as grazing, weed invasions, disease and pests, and changes in land use could drastically alter the response of the vegetation to climatic changes.

What Changes in Forest Productivity Will Occur as Measured by Changes in Volume, Growth, and Biomass?

What Are the Potential Impacts on the Forest Sector Under Climate Change, as Measured by Employment and Timber Prices?

When Forest Policy Questions for the RPA Assessment, Such as Reduced NFS Harvest, Are Examined With and Without Climate Change, Do the Forest Sector Impacts Differ Greatly in Magnitude or Kind?

Mills et al. (this volume) describe the use of two forest sector models (TAMM and FASOM) to estimate the effects of climate change on forests. Both models discussed in this chapter have been used to examine a variety of policy and management scenarios, including investment behavior of private owners, linkages between forestry and agriculture, and impact of altered trade flows on the forest sector globally. Both models share a number of characteristics in modeling timber demand and supply but they differ in their selection criteria for land management. Use of both the TAMM and FASOM models reveals the differences between: 1) likely future paths for the forest sector if historical relationships between key variables continue; versus 2) production possibilities and optimal responses to external events (e.g., climate change) and policies. FASOM assumes perfect knowledge of the future and optimal adjustments in the unfettered case. Under a climate change scenario, FASOM shows a greater shift to pine plantations, and as timber prices fall, an increase in land moving from forest to agriculture and a decrease of investment in pine plantations.

Both models challenge ecologists and policy analysts to be explicit in the size, location, and timing of various impacts, to consider the transition from current vegetation, and to gauge the tradeoffs between near-term policy concerns and long-term ecological impacts. Both models offer a common framework for integrating biophysical and social systems and for tracing how changes in typically biophysical attributes (growth, area of certain types, etc.) affect various measures of economic benefits and costs. In that role, these models operate at the interface of science and policy where the emphasis is on how models improve the information available for decision makers; that same information from the policy perspective helps shape perceptions about the effectiveness of various management actions.

What Are the Opportunities and Costs of Emissions Mitigation Using Forest Ecosystem Management and Forest Products Technologies?

Increasing the amount of carbon that is stored in forests could mitigate the emissions of carbon dioxide from fossil fuel combustion. Quantifying the amount of carbon that could be stored and the amount of carbon that is removed through harvest is critical in evaluating the opportunities and costs of this type of mitigation. The carbon budget approach uses inventory and field research data to compute the exchange of carbon between forests and the atmosphere. When linked to timber inventory models, these budgets can be used to analyze the impacts of alternative policy considerations on carbon storage. When linked to forest sector models, these budgets can facilitate an examination of the opportunities and costs using forest management to mitigate carbon emissions.

Heath and Smith (this volume) examine two accounting systems for soil carbon: 1) the accounting framework used by Birdsey and Heath in the 1993 RPA assessment, and 2) the framework presented in the National Greenhouse Gas Inventories of the International Panel on Climate Change (IPCC). Both systems base predictions on forest inventory variables of volume and area. Both methods recognize the importance of previous land use. The IPCC default system explicitly counts soil carbon loss when forests are cleared and cultivated, but does not include the accumulation of soil carbon due to afforestation (although soil carbon increases due to differing agricultural tillage practices are included). Birdsey and Heath explicitly account for the accumulation of soil carbon due to afforestation but do not explicitly count soil loss after deforestation. This is because the RPA analysis focused only on carbon in the forest sector. Deforested areas were assumed counted in the agricultural or urban sector, not forests, and over the last 30-40 years more land has become afforested than deforested.

Recent scientific studies indicate that harvesting may influence soil carbon, an initial slight increase followed by a decrease, and finally an increase. Heath and Smith (this volume) speculate that soil carbon will eventually return to pre-harvest levels. This corresponds to the pattern in the soil carbon assumptions in the RPA analysis. The magnitude of the effect seems to depend on the level and type of disturbance from logging operations. Consideration of the overall level of disturbance in harvesting operations in countries with active forest management could be used to revise soil carbon assumptions accordingly. Soil carbon decreases for 20-30 years following deforestation and cultivation and then remains relatively constant; following afforestation, soil carbon increases at a more gradual rate than the rate at which it had decreased, eventually becoming somewhat stable.

Smith and Heath (this volume) present some useful considerations for interpreting and using information for probabilistic assessments of uncertainty. They emphasize the consequences of summarizing uncertainty as well as how such summaries can affect the perception of uncertainty in subsequent use of the information. Examples are presented from their current forest carbon budget modeling efforts where they employ probabilistic definitions of uncertainty in Monte Carlo simulations.

Skog and Nicholson (this volume) provide historical estimates and projections of carbon sequestered in wood and paper products in the United States, and compare them to amounts sequestered in U.S. forests. The stocks of carbon in forests, in wood and paper products in use, and in dumps and landfills are large and increasing. Since 1910, an estimated 2.1 Pg (billion metric tons) of carbon have accumulated and currently reside in wood and paper products in use and in dumps and landfills, including net imports. For purposes of comparison, the current inventory of carbon in forest trees is 13.8 Pg and in forest understory, floor, and soils, 24.3 Pg. Net sequestration is computed as net imports of wood and wood products minus the emissions from decay and burning each year. Net sequestration of carbon in U.S. wood and paper products is projected to increase from 59 Tg/year in 1990 to 74 Tg/year by 2040, while net sequestration to forests is projected to decrease from 274 to 161 Tg/year (Tg is 1 million metric tons). Net sequestration is increasing in products and landfills because of an increase in wood consumption and a decrease in decay in landfills compared with phased-out dumps. Annual net sequestration to forest, product, and landfill carbon stocks is slightly greater than annual removal of carbon from the atmosphere by U.S. activities. Forest, product, and landfill stocks were 333 Tg in 1990, while net sequestration to forests, products, and landfills was 331 Tg in 1990. This difference is because net sequestration to stocks include net imports while annual net removals from the atmosphere does not. The choice of accounting influences the carbon storage results and is critical in decisions about which countries will receive credit for sequestering carbon.

Birdsey et al. (this volume) conclude that the potential for increasing carbon storage in forests in the United States is quite large. Potential carbon storage is governed by the biological potential of forest land to maintain biomass, the availability of suitable land for forests, and the costs and tradeoffs associated with increasing and maintaining (protecting) a higher level of carbon in forests. Although it is practically impossible to maintain all forests at maximum growth and carbon storage simultaneously, there is a biological and economic potential to increase growth rates and the amount of carbon stored.

Projections indicate that even without a forest carbon program, substantial increases in forest carbon are likely consequences of current timber market activities and forestry policies. There is some uncertainty over time, especially if climate change impacts on ecosystems are substantial and cause catastrophic reductions in biomass as forest ecosystems attempt to adapt. The storage of carbon in forests is generally considered a short-term activity because of these limits. But to the extent that reductions are needed sooner rather than later, forestry actions could be an integral part of any comprehensive greenhouse gas reduction strategy.

Birdsey et al. (this volume) echo Skog and Nicholson (this volume) in identifying the importance of the amount of carbon stored in wood products (in use or permanent disposal). Birdsey et al. (this volume) note that it is also possible to reduce greenhouse gas emissions from the forest sector by increasing energy efficiency in converting timber to products.

Size of programs, geographic location, and cost estimates vary widely because of differences in how past behavior is considered, differences in carbon accounting, and differences in model parameters. Considering costs and potential impacts, and recognizing that some options have not been analyzed sufficiently, "improved forest management" appears to offer the most cost-effective means to sequester additional carbon in forest ecosystems in the short term. Verification of carbon changes attributable to forest management may be difficult because we lack sufficient experimental research that quantifies impacts of specific practices on different carbon pools.

Afforestation costs are high relative to reforestation, but considering the uncertainty of the estimation process and the fact that costs/ton increase as afforestation programs expand, some program level less than about 20 million acres could be cost-effective. Afforestation may also be needed to offset conversion of forest land to other uses (deforestation). The potential of afforestation is limited primarily by the availability of suitable land (for ecological or economic reasons), nursery capacity, willingness of landowners to participate, and availability of technical assistance.

Use of biomass energy will also be important, although we do not have good cost/benefit estimates available at this time. Some simulations have shown that biomass-fueled power is not very competitive with coal without subsidies. Substitution of wood products for other energy-intensive materials may also be effective, but estimating and attributing the benefits are difficult. Urban tree planting and energy efficiency in wood product manufacturing will both be important factors.

Mitigation options can be analyzed most effectively within the context of the broad array of land use dynamics and forest cover type changes that are driven by other factors besides forest carbon considerations. Possible unintended consequences of carbon sequestration policies warrant close attention by those formulating policies. Important considerations are possible effects on other sectors of the economy for large-scale and concentrated afforestation efforts, timing of carbon impacts from deforestation versus longer-term afforestation, and uncertainties in climate change projections.

Mitigation policies cannot be evaluated independently of behavioral, economic, and institutional adjustments engendered by changing climate, both in the forestry and agriculture sectors. For example, if some agricultural producers respond to climate change by increasing the amount of land under cultivation, the amount of land available for forest carbon sequestration could be reduced. Within the forestry sector, producers may attempt to adapt to climate change by adopting appropriate tree planting mixes and practices. Further, increased research and technology transfer could promote technical advances that could help forest growers adjust to soil or other climatic characteristics. Long-run projections indicate that adaptations through forest carbon programs may not necessarily involve land use and forest management changes in a smooth or regular fashion over time, and that land use shifts to meet policy targets need not be permanent.

A number of policy tools involving forestry actions are available, including slowing the loss of forest land (deforestation) to urban and developed uses and agriculture. Mitigation policies involving increases in forest carbon should be formulated with an awareness that a substantial increment to the U.S. population is projected to be added over the next several decades. Such population increases are likely to increase pressure to develop additional forest land. Specific mixes of mitigation activities could be analyzed when concrete policy targets are developed.

Overview: Assessing the Impacts of Climate Change on U.S. Forests

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Introduction

The increasing concentration of atmospheric carbon dioxide has raised concerns about the vulnerability of forests to potential changes in climate and climate variability. These concerns have prompted governments around the world to commission technical assessments on the impact of climate change on the environment and the economy. Based on the current scientific information within these assessments, governments have initiated negotiations on policy actions to reduce greenhouse gas emissions and to address the vulnerabilities of the ecological, economic, and social systems to climate change. Critical to policy formulation is a periodic synthesis of the ever-expanding knowledge on forest ecology, the impact of climate on forests and of forests on climate, forest management, the socio-economic value of trees and forests, and the role of forests in the global carbon cycle.

The Forest Service conducts periodic assessments of the condition of forest and rangeland resources under the authority of the Renewable Resources Planning Act (RPA). The structure of these periodic assessments allows for the synthesis and integration of the current state of scientific knowledge (U.S. Department of Agriculture Forest Service 1989, 1994). As part of the RPA process, this report is a synthesis of current information that assesses the impact of climate change on U.S. forests. Policy questions critical to understanding the impact of global climate change on current and future trends (Joyce et al. 1997) form the basis for the subsequent chapters in this report. This chapter describes the synthesis of scientific information and assessment of the impacts of climate on forests, current understanding of the global climate, and the policy questions addressed in this assessment.

The Synthesis of Scientific Information

International Syntheses

Mandates to synthesize scientific information for policy formulation have developed from international organizations, international agreements between countries, and laws within countries (fig. 1.1). Internationally, countries have worked together to organize the scientific community to study the impact of climate change on the climate system, global ecosystems, and social and economic systems. Within the United States, a series of laws have mandated these assessments, which in turn have supplied information to international efforts. To provide context, we introduce this chapter by describing the development of international and U.S. assessments on climate change.

The United Nations Environmental Programme and the World Meteorological Organization established the Intergovernmental Panel on Climate Change (IPCC) in 1988 in order to: 1) assess available scientific information on climate change; 2) assess the environmental and socio-economic impacts of climate change; and 3) formulate response strategies. The first assessment reports were completed in 1990 (Houghton et al. 1990; IPCC 1991), the second reports were completed in 1995 (Bruce et al. 1996; Houghton et al. 1996; Watson et al. 1996), and the third report is being written. These recent IPCC assessments have identified the importance of integrating the ecological and the economic and social analyses (Houghton et al. 1996; Bruce et al. 1996) to develop policy direction for mitigation and adaptation to an increasingly changing climate. The third assessment will rely on country assessments such as the U.S. assessments where the analysis can focus more closely on the impact of climate change on individual countries.

In 1992, the United States and over 50 other nations signed the Framework Convention on Climate Change (FCCC), an international agreement with no binding obligations. The policy objective identified in the FCCC was to achieve "stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system." In addition, these countries agreed that "such a level should be achieved within a time-frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened and to enable economic development to proceed in a sustainable manner." The signing of this agreement initiated a series of international meetings, so-called Conference of the Parties (COP), at which negotiators determine the mechanisms by which greenhouse gas concentrations could be stabilized globally (fig. 1.1).

After signing the FCCC, the United States developed policy and preferred actions to stabilize U.S. emissions by the year 2000 at the 1990 levels (Clinton and Gore 1993; U.S. Dept. of Energy 1994). Strategies within the Climate Change Action Plan included emission-reducing activities within the transportation and manufacturing sectors of the economy, and carbon storage activities in the forest

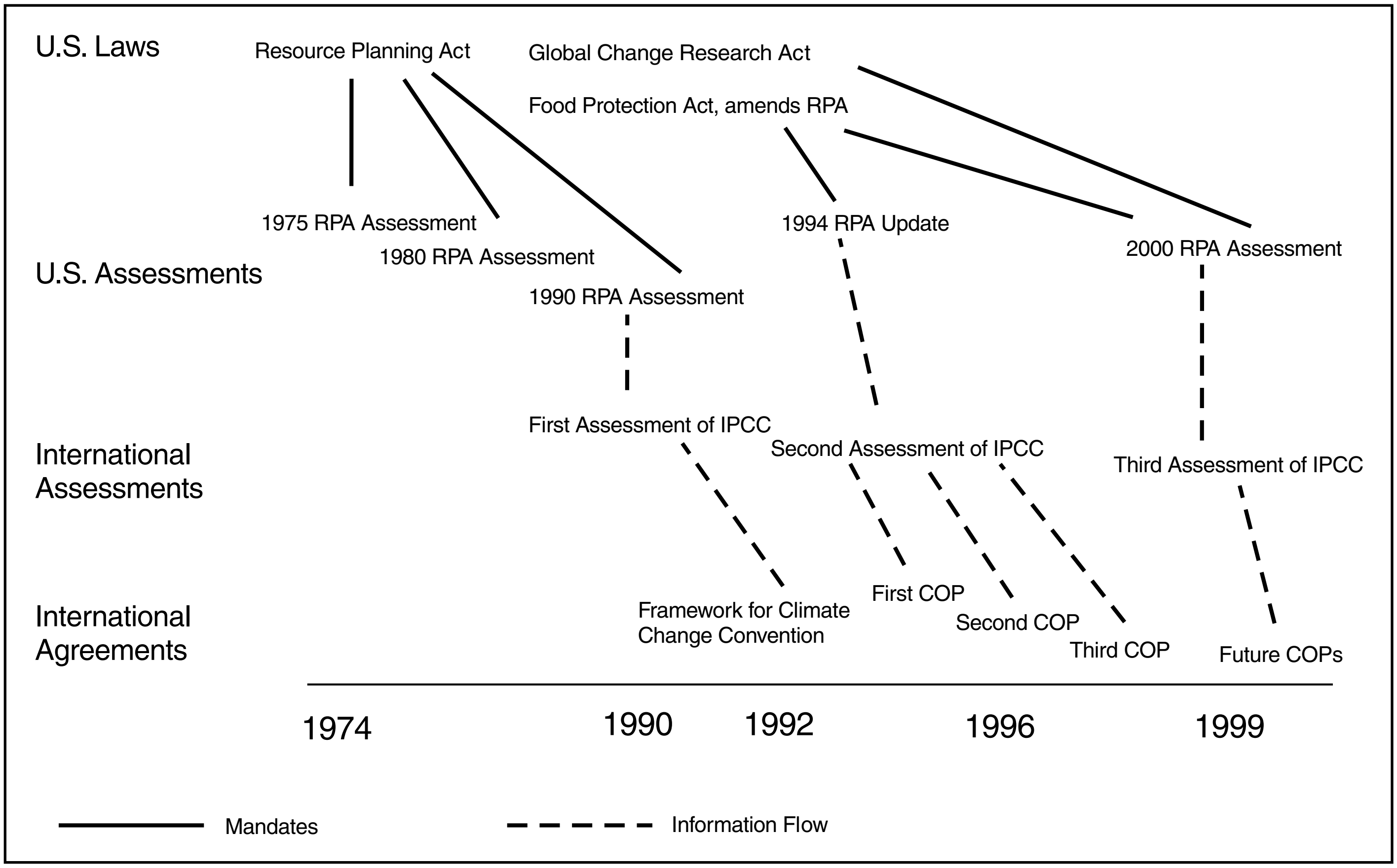

Figure 1.1—Laws and International Agreements mandating climate change assessments in the United States and internationally.

sector. The forest sector currently sequesters more carbon than it emits, and there are opportunities to increase this offset of fossil fuel emissions in the near-term. The proposed activities included accelerating tree planting and encouraging forest management evaluation in non-industrial private forests. These carbon-storage activities would allow time to develop ways to reduce fossil fuel emissions.

A discussion on the importance of greenhouse gas stabilization led countries at the Third Conference of the Parties, held in Kyoto, Japan, in December 1997, to produce an agreement that included binding targets for reducing emissions and flexible implementation where targets would vary by country and groups of countries. Under the terms of the agreement, which has not yet been ratified by the U.S. Senate, the U.S. is bound to reduce emissions 7% below 1990 levels by 2008-2012. This reduction by 2012 is substantial, given that increases in population and economic expansion would increase future emissions in the absence of controls. Only reduction activities initiated in 1990 or later may be counted, since this is the reference point against which all future changes will be measured. These discussions included the role of forestry and land use change in stabilizing and mitigating carbon emissions. Negotiators considered the potentially important role of forest management in the ability of the United States to meet its binding targets of greenhouse gas emissions; yet, it is still not clear whether forest management will be included.

The importance of forests in maintaining the global carbon cycle was recognized formally for temperate and boreal forests in the Santiago Declaration, a statement signed in 1995 by the governments of Australia, Canada, Chile, China, Japan, the Republic of Korea, Mexico, New Zealand, the Russian Federation, and the United States. This statement identifies a comprehensive set of criteria and indicators for forest conservation and sustainable management for use by government policy makers. A criterion is a category of conditions or processes by which sustainable forest management may be assessed, and it is characterized by a set of related indicators that are monitored periodically to assess change. The United States is implementing many of these criteria and indicators within forest inventory and monitoring programs nationally. Criterion 5 is the maintenance of forest contribution to global carbon cycles.

U.S. Laws and the Forest Service Resource Assessments

The Forest and Rangeland Renewable Resources Planning Act of 1974 directed the Secretary of Agriculture to prepare a Renewable Resources Assessment in 1975 and a decadal update starting in 1979. The assessment was to include "an analysis of present and anticipated uses, demand for, and supply of the renewable resources, with consideration of the international resource situation, and an emphasis of pertinent supply, demand and price relationships trends." Since 1974, there have been 3 national assessments and two updates which have reviewed the current and likely future condition of forest and range resources including wildlife, water, timber, recreation, range forage, and minerals. Assessments typically include: 1) description of the current status of the resource; 2) a projection of supply of and demand for resource outputs; 3) social, economic, and environmental implications of the projections; 4) management opportunities to improve the resource situation; and 5) a description of Forest Service programs and responsibilities. The results of the RPA assessment are used as the factual basis for formulating future renewable resource management programs. The structure of these on-going assessments provides a mechanism by which current scientific information can also be synthesized periodically to address policy questions.

Subsequent laws within the United States mandated assessments of the impact of climate change on the U.S. environment and economy (fig. 1.1). The Global Change Research Act of 1990 requires the National Science and Technology Council to: 1) assess current human-induced and natural trends in global change; 2) analyze effects of global change on the natural environment, agriculture, energy production and use, land and water resources, transportation, human health and welfare, human social systems, and biological diversity; and 3) project major trends for the subsequent 25 to 100 years. The 1990 Food Protection Act amends the 1974 Resources Planning Act and requires the Forest Service to: 1) assess the impact of climate change on the condition of renewable resources on forests and rangelands, and 2) identify the rural and urban forestry opportunities to mitigate the buildup of atmospheric carbon dioxide.

Since the Amendment of the RPA, the RPA assessments have included an analysis on the vulnerability of U.S. ecosystems to changes in climate, and the potential impact on the social and economic systems from changes in climate. The 1989 assessment included a review of the current scientific understanding of the potential effects of global climate change on forests (Joyce et al. 1990). The next assessment update in 1993 used an integrated modeling framework to analyze the impact of climate change on ecosystem productivity, timber supply and demand, and carbon storage (Joyce 1995; Joyce et al. 1995). We use this modeling framework to structure our current synthesis of the impact of climate change on U.S. forests (fig. 1.2). The following chapters review our ability to quantify the impacts of a changing climate on changes in vegetation communities (Chapter 2), forest productivity (Chapter 3), forest economy, land area, timber inventory (Chapters 4 and 5), and carbon stored in forests, in wood products, and in landfills and dumps (Chapters 5, 6, 7 and 8).

Six policy questions related to the impact of global climate change on forests (Joyce et al. 1997) form the basis for the subsequent chapters in this report. This chapter describes mandates and structures of synthesizing scientific information and assessing the impacts of climate on the forest sector, current understandings of the global climate, and policy questions addressed in this assessment.

Understanding the Dynamics of Climate

Climate Dynamics, Greenhouse Gases, and Global Carbon Cycle

Identifying the vulnerabilities of ecosystems and economies to climate variability and change depends on an understanding of the sensitivity of those systems to climate. Analyzing the effectiveness of policy instruments in stabilizing greenhouse gases, such as sequestering carbon in forests, depends on an understanding of several factors: climate processes, the physical changes in climate arising from all greenhouse gases and aerosols, biospheric and oceanic interactions, and the influence of humans on climate processes and forest biogeochemistry through activities such as forest management and land use change. We briefly review current observations on changes in atmospheric chemistry, changes in global and U.S. climates, and the influence of humans on the earth's climate system.

Certain atmospheric gases have the potential to warm the atmosphere and are collectively known as greenhouse gases: carbon dioxide, methane, nitrous oxides, chlorofluorocarbons, and water vapor (Houghton et al. 1996). The amount of warming is a function of the ability of these gases to absorb solar radiation (radiative properties of the gases) and the atmospheric concentration of each gas. The radiative property of a gas is constant, but the atmospheric concentrations of these gases are altered by natural processes and human activities. It is the rise in atmospheric concentration of these gases that is of concern globally.

The atmospheric concentration of carbon dioxide, methane, nitrous oxides, and the chlorofluorocarbons has

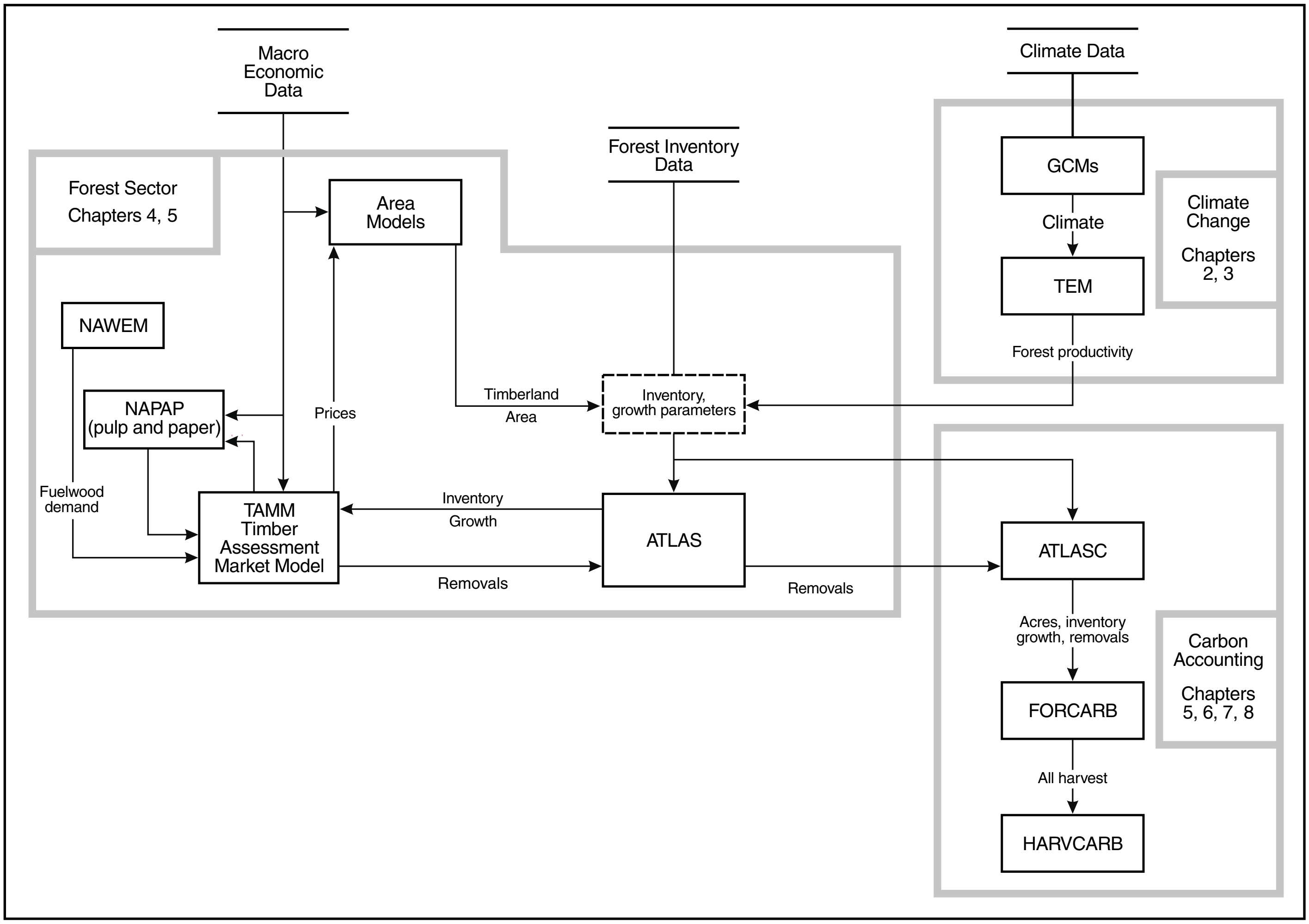

Figure 1.2—Components of the Integrated Modeling system used in the 1993 Forest Service RPA Assessment Update.

increased since pre-industrial times (table 1.1). Increases range from 13 percent for nitrous oxides to 145 percent for methane. Moreover, the atmosphere did not contain chlorofluorocarbons in pre-industrial times. Increases in carbon dioxide are mainly the result of fossil fuel emissions from industrial and domestic activities and land use conversions. Methane increases result from the production and use of fossil fuel and from anthropogenic activities such as rice cultivation and livestock production. The sources of nitrous oxides are small and hard to quantify, but include agriculture and industrial processes. The rates of concentration changes (table 1.1) are positive except for CFC-11, which is being controlled as a result of the Montreal Protocol. The positive rates of change demonstrate that atmospheric concentrations will continue to increase for these greenhouse gases, unless the activities influencing these concentrations are modified.

While concentrations of greenhouse gases are sources of atmospheric warming, other processes have recently been identified that also influence the earth's energy. Aerosols, tiny particles of liquid or solid matter suspended in the atmosphere, can be derived from many different materials including sea salt, soil, smoke, and sulfuric acid (Schimel et al. 1996). They increase the scatter of incoming solar radiation, sending some radiation away from earth. They are also a part of the cloud-forming process. In both of these ways, aerosols can influence the earth's temperature. The length of time that aerosols remain in the atmosphere is much less (a few weeks) than the residence time of carbon dioxide (approximately 100 years). In addition, human-produced aerosols do not mix throughout the globe like carbon dioxide (Charlson et al. 1992). They tend to remain near the area of generation and thereby have an impact on the regional climate.

Land management activities influence the uptake and release of greenhouse gases. The processes that influence these carbon fluxes operate at different spatial scales and time frames. Currently, the main sources of carbon dioxide include fossil fuel consumption and land use change, particularly deforestation in the tropics. The main res-

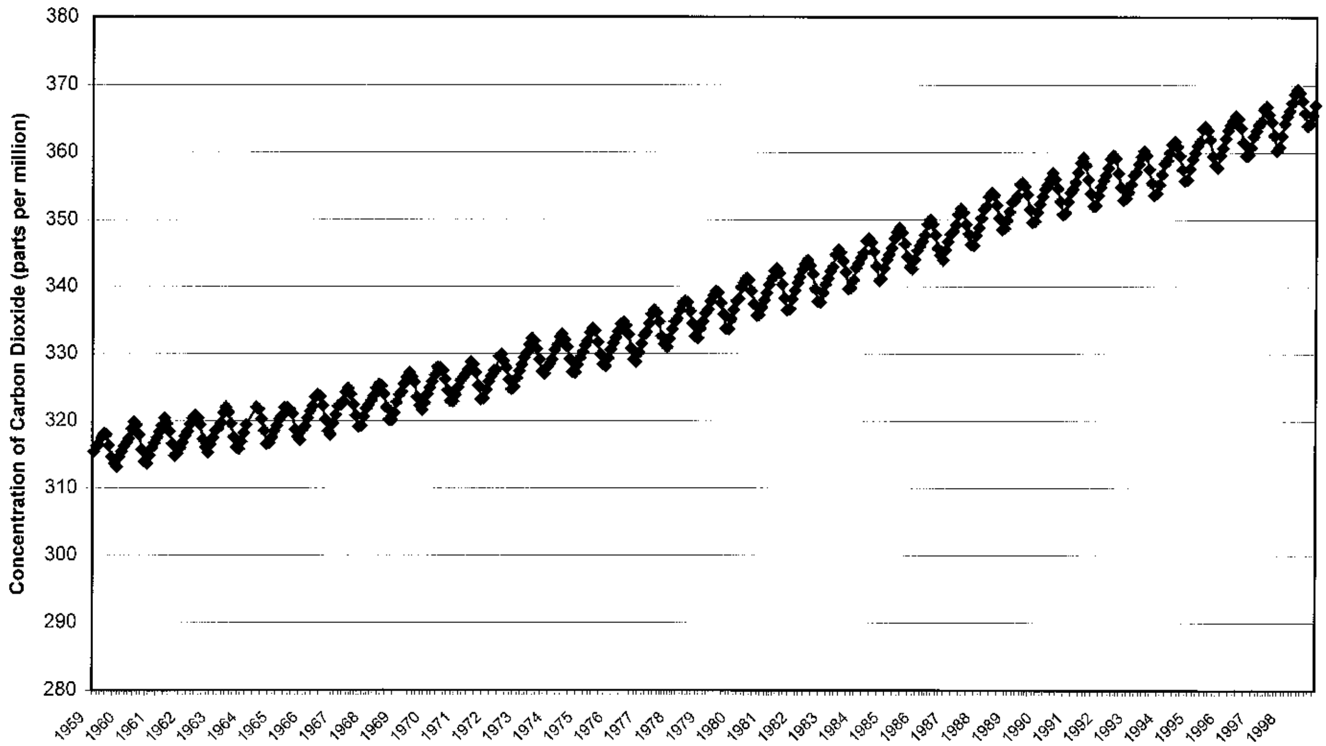

Figure 1.3—Observed concentration of atmospheric carbon dioxide in parts per million at Mauna Loa, Hawaii, from 1959 to 1998. (Source: C. D. Keeling, Scripps Institution of Oceanography)

Table 1.1—A sample of greenhouse gases affected by human activities (Houghton et al. 1996).

	CO_2	CH_4	N_2O	CRC-11	HCFC-22 (A CFC substitute)	CH_4 (A perfluorocarbon)
Pre-industrial concentration	~280 ppmv	~700 ppbv	~275 ppbv	zero	zero	zero
Concentration in 1994	358 ppmv	1720 ppbv	312[1] ppbv	268[1] pptv[2]	110 pptv	72[1] pptv
Rate of concentration change[3]	1.5 ppmv/yr	10 ppbv/yr	0.8 ppbv/yr	0 pptv/yr	5 pptv/yr	1.2 pptv/yr
	0.4%/yr	0.6%/yr	0.25%/yr	0%/yr	5%/yr	2%/yr
Atmospheric lifetime (years)	50–200[4]	12[5]	120	50	12	50,000

[1] Estimated from 1992–93 data.
[2] 1 pptv = 1 part per trillion (million million) by volume.
[3] The growth rates of CO_2, CH_4, and N_2O are averaged over the decade beginning 1984; halocarbon growth rates are based on recent years (1990s).
[4] No single lifetime for CO_2 can be defined because of the different rates of uptake by different sink processes.
[5] This has been defined as an adjustment time which takes into account the indirect effect of methane on its own lifetime.

ervoirs for carbon storage include the atmosphere, the ocean, and the vegetation. Incorporation of carbon into vegetation is the fastest process, and atmospheric concentrations throughout the year reflect the seasonal growth of vegetation (fig. 1.3). Transfers to soils and ocean depths operate on the decade-to-century time-scale.

Transfers of carbon dioxide between the atmosphere, ocean, and land at the global scale have been examined using a budgeting approach (Houghton et al. 1996). The amount of carbon dioxide that remains in the atmosphere is used to project likely future changes in the global climate. Emissions from fossil fuel combustion and cement production are the larger share of the carbon sources identified (table 1.2). Atmospheric sampling and forest inventories indicate that the carbon source of land clearing in the tropics is approximately balanced by the carbon reservoir of forest regrowth in the Northern Hemisphere. Experimental research suggests that the uptake of carbon in vegetation may be stimulated by increased atmospheric carbon dioxide and nitrogen fertilization from the deposition of nitrogen in the atmosphere (Kauppi et al. 1992, Aber et al. 1998, Magill et al. 1997). The future role of vegetation in the global budget is highly uncertain because of our lack of understanding about processes such as fertilization from atmospheric carbon dioxide and our inability to predict future rates of deforestation in the tropics and regrowth in the mid-latitudes (Houghton et al. 1996, Watson et al. 2000). Understanding the uptake and release of carbon in forested ecosystems, especially as affected by management activities, will be important in addressing the role of forestry, not only in mitigating greenhouse gas emissions, but also in the processes influencing the global carbon budget (Schimel et al. 2000).

Analyses in the recent RPA Update focused on the role of forestry in releasing carbon through harvest and in storing carbon through growth and land conversion to forests on the U.S. mainland. The net effect of these activities in the United States comprise an estimated carbon sink of approximately 0.3 GtC/yr (Birdsey and Heath 1995), a substantial portion of the total uptake by the Northern Hemisphere. These analyses are set in the context of the global budget of carbon in order to determine what role U.S. forests might play in mitigating carbon emissions and thereby to help stabilize the concentrations of carbon dioxide in the atmosphere.

Table 1.2—Annual average anthropogenic carbon budget for 1980 to 1989. CO_2 sources, sinks, and storage in the atmosphere are expressed in GtC/yr (where GtC is gigatons of carbon) (Houghton et al. 1996).

CO_2 sources	
(1) Emissions from fossil fuel combustion and cement production	5.5 ± 0.5[1]
(2) Net emissions from changes in tropical land-use	1.6 ± 1.0[2]
(3) Total anthropogenic emissions = (1) + (2)	7.1 ± 1.1
Partitioning amongst reservoirs	
(4) Storage in the atmosphere	3.3 ± 0.2
(5) Ocean uptake	2.0 ± 0.8
(6) Uptake by Northern Hemisphere forest regrowth	0.5 ± 0.5[3]
(7) Inferred sink: 3-(4+5+6)	1.3 ± 1.5[4]

[1] For comparison, emissions in 1994 were 6.1 GtC/yr.

[2] Consistent with Chapter 24 of IPCC Working Group II (Watson et al. 1996).

[3] This number is consistent with the independent estimate, given in IPCC Working Group II (Watson et al. 1996), of 0.7 ± 0.2 GtC/yr for the mid-land high latitude forest sink.

[4] This inferred sink is consistent with independent estimates, given in Chapter 9 of IPCC Working Group I (Houghton et al. 1996), of carbon uptake due to nitrogen fertilization (0.5 ± 1.0 GtC/yr), plus the range of other uptakes (0-2 GtC/yr) due to CO_2 fertilization and climatic effects.

Observed Trends in Climate at the Global Scale

At the global scale, increases in air temperature and in precipitation have been documented in the historical record of observation (Houghton et al. 1996). Both sea surface and land surface temperatures indicate a warming pattern. While observed changes related to temperature generally have a higher confidence than observed changes in the hydrological cycle, precipitation has also increased 1% globally.

Since the late 19th century, near-surface air temperatures have risen from 0.3 to 0.6°C, paralleling similar increases seen in near-surface ocean temperatures. The most reliable period of observation, the last 40 years, indicates a warming of 0.2 to 0.3°C for the global average surface temperature (Houghton et al. 1996). While temperatures have increased over time in urban centers, the increases in urban temperatures and the expansion of urban areas contributes minimally to global surface warming (Easterling et al. 1997). Urbanization may be important in some regions, however. Similarly, desertification has influenced local climates, but has a negligible effect on global temperature changes (Houghton et al. 1996).

The difference between the surface maximum and minimum daily temperatures has decreased since the middle of the 20th century based on an analysis of over 54 percent of the global land area (Easterling et al. 1997). This narrowing of the daily maximum and minimum temperatures is the result of warmer nighttime temperatures, which may reflect not only the increase of carbon dioxide but also increased cloud cover. Daytime clouds obstruct the daytime sunshine, while nighttime clouds reduce the amount of terrestrial radiation escaping at night.

A number of indirect indicators support these observed increases in temperature globally. The 20th century retreat of mountain glaciers and the underground temperatures in boreholes are seen as indirect indicators supporting these warming estimates. Houghton et al. (1996) reported mass balance declines for the six glaciers for which long observational data are available. South Cascade in Alaska showed the largest loss in mass balance. Underground temperatures in boreholes have been observed to warm in New England, Canada, Alaska, France, and the ice sheet in the Arctic regions, but other areas have shown no changes. An analysis of all the North American studies concluded that underground temperatures warmed between 0.3 and 4.0 ^{0}C since the 19th century (Deming 1995). The increasing trends in precipitation have also been corroborated regionally with indirect indicators such as streamflow, lake levels, and where available, soil moisture.

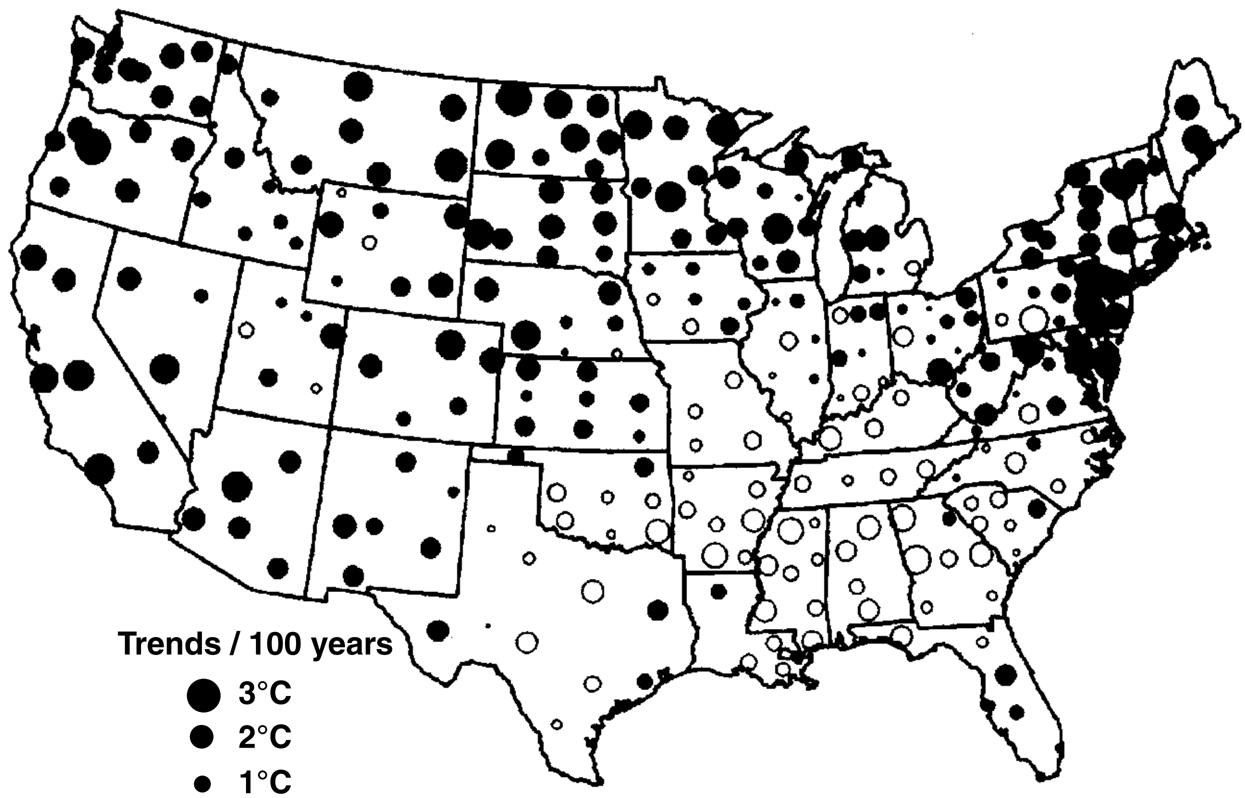

Figure 1.4—Temperature trends (1900-94 converted to mean temperature in °C per 100 years) centered within state climatic divisions are reflected by the diameter of the circle centered within each climatic division. Solid circles represent increases and open circles, decreases (from Karl et al. 1996).

The variability of climate is calculated from the historical records. Globally, the data are inadequate to assess whether climate variability has changed in response to elevated greenhouse gases (Houghton et al. 1996). No global-scale patterns in drought frequency or intensity or variation in rainfall events or extremes has emerged from the analysis of the available data. Sufficient data have been available to examine these trends for some regions, such as described below for the United States.

Indicators of Change in the U.S. Climate

An analysis of the near-surface air temperature reveals that temperatures have warmed over much of the United States in the last 100 years (fig. 1.4) (Karl et al. 1996). Temperature trends at the national scale, if represented with a linear trend, indicate a rise of about 0.4°C over 100 years. This rise occurs mainly in the first six months of the year. Regional records show the South with a slight cooling (1°C/100 years) and the northeast, northcentral, and western parts of the United States with a warming trend of 1 to 2°C. At the continental scale, Watson et al. (1998) reported the highest increases in warming occurred along an area extending from northwestern Canada across the southern Canada/northern U.S. region to southeastern Canada and the northeastern United States. The temporal pattern of these increases indicates an increase in warming from the 1920s to the 1940s and again from the 1970s to the 1990s.

Within the United States, precipitation was shown to have increased since 1970 about 5%, mainly the result of increases in precipitation in the last six months of the year, and primarily in autumn (Karl et al. 1996). The largest increases, up to 20%, were seen in the Gulf Coast states, the lower northeastern part of the United States, and the midwestern states (fig. 1.5). However, states such as California, Montana, Wyoming, North Dakota, parts of Colorado, and Nebraska have actually had a decrease in annual precipitation of similar magnitude.

Karl et al. (1996) present a framework for examining potential changes in the U.S. climate. They developed two indices that reflect the behavior of individual climate metrics that would likely reflect changes in the climate as a result of increasing concentrations of greenhouse gases. Their Climate Extremes Index supports the notion that the climate of the United States has become more extreme in recent decades. Their U.S. Greenhouse Climate

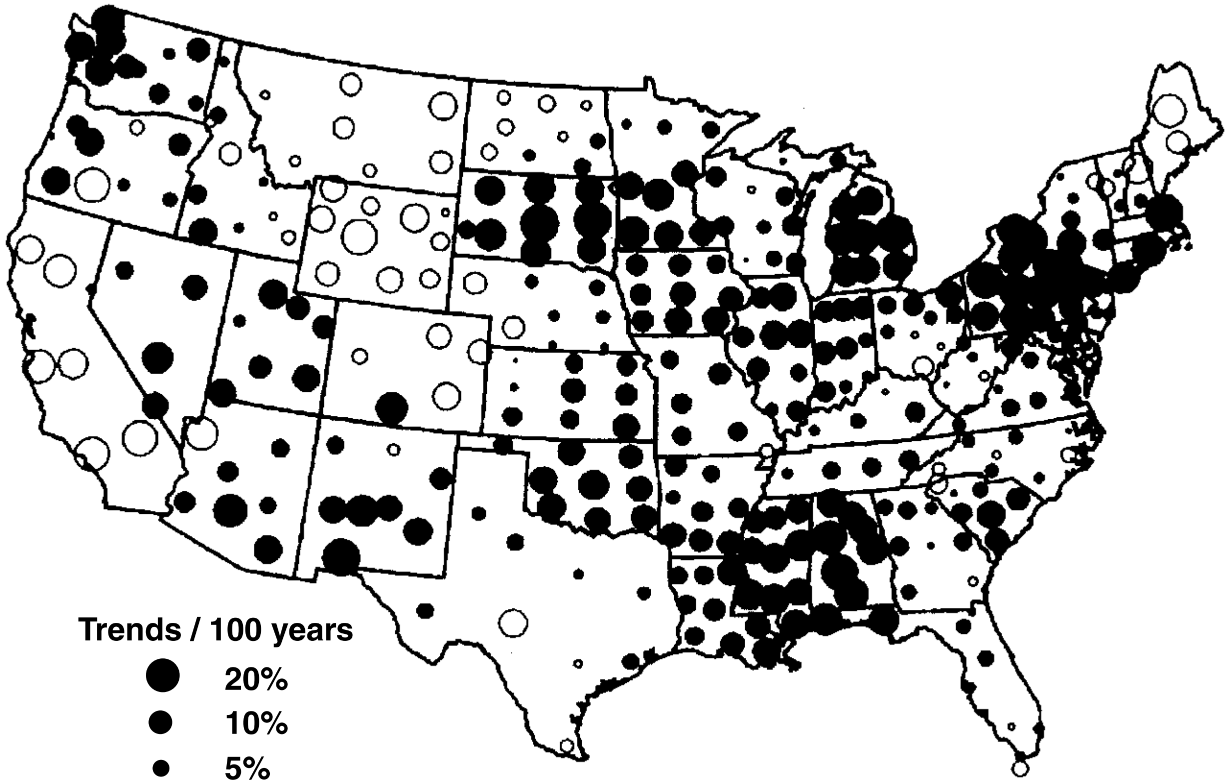

Figure 1.5—Precipitation trends (1900-94 converted to percent per century) centered within state climatic divisions are reflected by the diameter of the circle centered within each climatic division. Solid circles represent increases and open circles, decreases (from Karl et al. 1996).

Response Index is consistent with an enhanced greenhouse effect. However, neither response is large enough to conclude that the increase in extremes reflects a nonstationary climate, or that the increase in the Greenhouse Climate Response Index may be the result of other factors including natural climate variability.

The increase in extremes is influenced markedly by three precipitation indicators: the frequency of long-term drought severity and moisture excess; the frequency of extreme 1-day precipitation events; and a much greater than normal number of days with precipitation. When Karl et al. (1996) analyzed the extremes associated with drought severity and moisture excess, they determined that there was considerable decadal variability in drought severity and in moisture surplus. The likelihood that these occurrences arose from a quasi-stationary climate was 25%. In the last several decades, however, they noted a tendency for more of the area in the United States to be either in a drought or to have severe excess moisture. Karl et al. (1996) determined that the proportion of the country that has had a much greater than normal amount of precipitation derived from extremely heavy (greater than 50.8 mm or 2 in) 1-day precipitation events could be reliably computed from climate data available since 1910 (fig. 1.6). They concluded that the steady increase in area of the United States affected by extreme precipitation events would be highly unlikely (less than 1 chance in 1000) in a quasi-stationary climate. The percentage of the conterminous U.S. area with the number of wet days much above normal also increased beyond what one would expect for a stationary climate. This increase in the number of wet days parallels the increase in precipitation at the national scale. The proportion of area in the United States with a much greater than normal number of dry days did not change over the century (Karl et al. 1996).

An increase, but of more recent nature, was seen in the percentage of the conterminous U.S. area with a much above normal cold season (October through April) precipitation (fig. 1.7). Here the increase is most noticeable since 1970. Another indicator of potential shifts was the decrease in area affected by much below normal maximum temperatures (not shown here).

Recent work has synthesized many climate metrics, including biologically meaningful indicators, to show a rapidly warming climate in Alaska. Chapman and Walsh (1993) documented a significant warming trend in the temperature records over the last few decades for most of Alaska, with winter temperatures warming more than

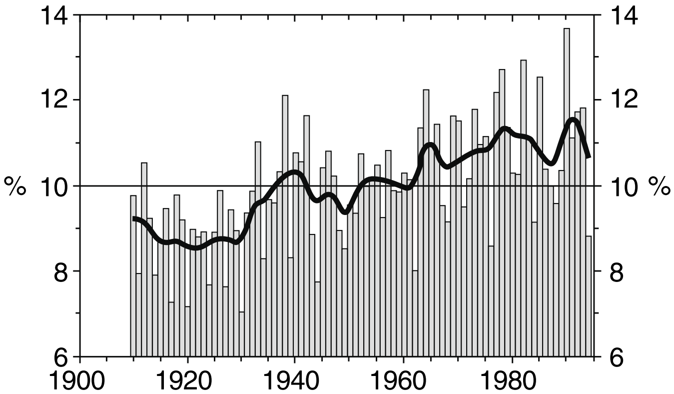

Figure 1.6—Percentage of the conterminous U.S. area with a much above normal proportion of total annual precipitation from 1-day extreme (more than 2 inches) events (from Karl et al. 1996).

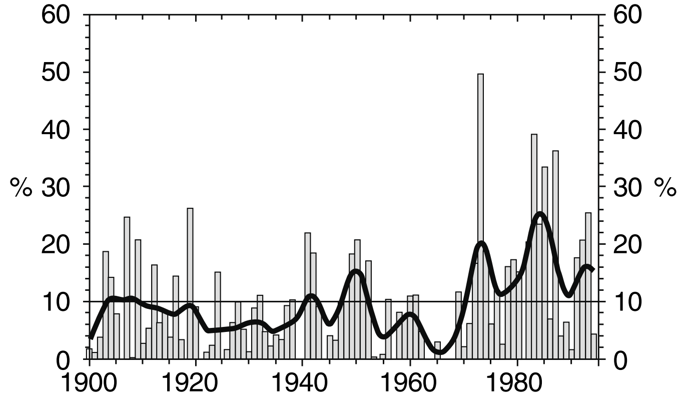

Figure 1.7—Percentage of the conterminous U.S. area with much above normal cold season (October through April) precipitation (from Karl et al. 1996).

summer temperatures. Jacoby et al. (1995) confirmed this recent trend by analyzing tree rings. They also concluded that temperatures are near the highest level of the past 3 centuries, an observation also made by Lachenbruch et al. (1988) from data derived from arctic boreholes. Most recently, Myneni et al. (1997) examined atmospheric CO_2 trends and changes in the normalized difference vegetation index (NDVI - an index of greenness). They concluded that the active growing season lengthened by about 12 days and that winter temperature increased by 4°C between 1981 and 1991 at latitudes above 45°N. Before this recent climate research, Oechel et al. (1993) reported changes in the carbon dioxide flux from Arctic tundra ecosystems, shifting the carbon balance from a net carbon dioxide sink to a source of carbon. This increase was presumed to be the result of increasing soil temperatures, soil aeration, and depth of soil thaw (Oechel et al. 1993).

Predicting Future Climates and the Vegetation Response

Atmospheric-Biospheric Relationships

Forests and climate are intimately connected in the United States. The North American climate is influenced by the region's size, topography, and the widely varying temperatures of the surrounding oceans. The current distribution of forests is strongly tied to these climate patterns.

In the Pacific Northwest of the United States, local climates are influenced by elevation, proximity to the Pacific Ocean, prevailing winds, and the north-south-oriented mountain ranges. Similarly, local climates on the eastern coast are influenced by proximity to the ocean and the Gulf of Mexico, and the episodic extreme events such as hurricanes. Even climates in the interior of the United States are influenced by large bodies of water such as the Northcentral communities surrounding the Great Lakes and the communities on the eastern side of the Great Salt Lake in Utah. For a large part of the North American continent, disturbances in the upper-level westerly winds play an important role in the temperature and moisture regimes. The Polar Front refers to the surface boundary between the colder, drier Arctic air and the warmer, moister air in the south. Disturbances in the upper level westerly winds shift the position of the upper level jet stream, and hence the Polar Front, back and forth across the North American continent. In the colder months of the year, this front moves slowly back and forth across the United States, bringing colder Arctic air to the northern and parts of the southern United States. Spring and fall see shorter, weaker systems moving quickly across the continent. In the summer the Polar Front retreats far into northern Canada. Because of these climate influences, the current temperature and precipitation gradients in the eastern half of the United States are strong in both the north to south and east to west directions.

Forests dominate the East and parts of the West. Major timber producing regions include the moist Pacific Northwest coast, the warm and moist Southeast, and the moist but cooler Northcentral region. Annual precipitation is highest along the Pacific Northwest coast and in the Southeast, centered mainly along the Gulf Coast states (Watson et al. 1998). High precipitation rates, low evaporative demands, and moderate temperatures characterize the Pacific Northwest climate (Lassoie et al. 1985). The forests in the East respond to climates influenced by proximity to the ocean and shifts in the continental air masses (Hick and Chabot 1985). Forests on the east coast periodically experience major tropical storms and hurricanes.

Any change in climate and climate variability has the potential to alter the structure, function, and geographic distribution of forests.

In the 1993 RPA Update, climate scenarios from four General Circulation Models (GCM) were used to examine the impact of climate change on forest productivity (Joyce et al. 1995). These global models provided equilibrium climates under elevated atmospheric carbon dioxide concentrations at a coarse spatial resolution. We review below the improved understanding of climate dynamics since this analysis. Another area where the understanding of climate dynamics has improved but there remains much uncertainty is the interaction between the land use and atmospheric dynamics. We also review below recent research identifying the contributions that land use makes to local climate conditions.

Improvements in Climate Scenarios

Since the development of these early GCMs, improvements have resulted in better depiction of large scale features of the climate system such as the seasonal, geographical, and vertical variations in climate (Houghton et al. 1996). Our ability to detect climate change is closely linked with our ability to predict the temporal and spatial variability of climate. Within the GCMs, the variability in results is broadly comparable to the observed variability in time and space (Houghton et al. 1996). Improved GCMs capture the relatively smaller variability over the oceans and the larger variability over continental interiors. However, only recently has the interannual variability associated with the El Nino-Southern Oscillation phenomenon been captured by a coupled atmospheric and ocean model, the Hadley GCM (Tett et al. 1997). The Hadley model and several other GCM models represent a significant improvement in the projection of climate change through the three-dimensional representation and interaction of atmospheric processes, oceanic processes, and the land surface properties on a time-dependent basis (Houghton et al. 1996). These scenarios are referred to as transient scenarios, in contrast to the earlier equilibrium scenarios. These computationally intensive simulations allow an examination of the behavior of climate as human-induced emissions increase over time.

While climate scenarios in the Second IPCC Assessment included the nature of change over time (Houghton et al. 1996), the IPCC analysis of the impact of climate change on ecosystems, including forests, was based on the earlier equilibrium climate scenarios. Only now is research being reported that has used the transient scenarios to examine the impact of climate change on forests (Neilson 1998). However, the land surface properties of these improved atmospheric-ocean coupled models is static; that is, the land surface properties, such as vegetation, do not change over time in response to climate changes or human activities. Recent work has shown the impact of land surface properties on climate modeling (Pitman et al. 1999). The development of feedbacks between land surface properties and the atmosphere-ocean processes is another area of needed research.

The addition of aerosols to the GCMs has resulted in closer agreement between model simulations and the observed global mean surface temperature. The release of stratospheric aerosols from the 1991 Mount Pinatubo eruption was used to exercise a climate model; the model results of a shift in the global temperature variation agreed closely with the observations. Analyses with the Hadley GCM indicate that the influence of aerosols varies by season and region of the globe (Mitchell and Johns 1997). In the winter, aerosols cool the warming influence of carbon dioxide; in the summer, the influence of carbon dioxide on the hydrological cycle is disrupted. Regional climates in Europe and Southeast Asia are significantly impacted by the inclusion of aerosols in the model.

These improvements in GCMs have been outpaced by an equally important increase in our understanding of the complexity of the climate system and the identification of additional processes that need to be included in the climate models. The range of temperature increases (1.5°C to 4.5°C) given in Houghton et al. (1990) and Houghton et al. (1996) in response to a doubling of carbon dioxide concentration results from model uncertainty associated with internal feedbacks such as water vapor feedback, cloud/radiative feedback, ice and snow albedo feedback, and uncertainties in the representation of ocean circulation and land-surface/atmosphere interactions.

Clouds influence the global temperature both as a cooling agent and as a warming agent. The formation of clouds is dependent upon the interactions of atmospheric water and aerosols. The uncertainty of the temperature rise is primarily the result of our lack of understanding of cloud processes. Sea ice coverage varies between GCMs and further refinement of this aspect will increase their accuracy (Houghton et al. 1996). Changes in the climate from anthropogenic emissions will influence environmental factors such as soil moisture, albedo, and vegetation. Changes in these surface properties will, in turn, affect the local climate.

GCMs typically operate at a coarse resolution. The complex topography of landscapes such as the western United States is not represented in detail in these GCMs. At regional scales, the interactions between the atmosphere and the surface (topography, vegetation) are important. The regional influence of human-generated aerosols will likely be significant as these aerosols do not disperse widely from their sources of generation. Further, most GCMs do not include changes in land use and these have been shown to have significant impact on temperature and precipitation changes, especially in the tropics and subtropics (Houghton et al. 1996).

Watson et al. (1998) concluded that limited confidence can be placed in regional climate projections because these projections are unable to capture present-day climates, and inter-model variability is quite large. Although statistical downscaling techniques and nested regional models have been used to refine regional climate projections, the current GCMs do not capture the complex topographical features, large lake systems, and narrow land masses that significantly affect regional and local change scenarios (Houghton et al. 1996). This degree of uncertainty complicates the assessment of the impact of climate and climate variability on forest resources at the local scale. Houghton et al. (1996) identified the following urgent scientific problems requiring attention: improved understanding of regional patterns of climate change including land-surface processes and their link to atmospheric processes; coupling of scale between global climate models and regional and smaller scale models; and simulations with higher resolution climate models.

Influence of Human-Induced Land Use Change on Climate

Land use change influences atmospheric-biospheric relationships (Cotton and Pielke 1995; Houghton et al. 1996) through changes in atmospheric chemistry and the surface characteristics such as albedo. The conversion of vegetation from forest to grassland, through harvest or burning, changes the roughness and albedo of the land surface, influencing the climate. Biomass burning is used to clear land for shifting cultivation, to convert land from forest to agriculture or grazing, to promote productivity of grasses or agricultural crops, and as an energy source (Crutzen and Andreae 1990). This burning produces trace gases and aerosol particles that influence atmospheric chemistry and climate. When tropical forests were replaced by pasture within the Amazon basin, mean surface temperature increased about 2.5°C and annual evapotranspiration decreased by 30% (Nobre et al. 1991). Two other effects observed in the model simulations, larger diurnal fluctuations of surface temperature and vapor pressure deficit, have been observed in deforested areas in the Amazon (Nobre et al. 1991).

The schemes used in GCMs to depict the land surface, including vegetation, have increased in their complexity since the first IPCC assessment, but there is still considerable uncertainty in their ability to predict soil moisture, surface heat, and water fluxes in the absence of land use changes. The slow changes in reforestation and the dynamic impacts of land use changes such as deforestation are not incorporated into the current GCMs (Houghton et al. 1996).

Large-scale changes in vegetation cover have resulted from deforestation to agriculture and reforestation in New England (Foster et al. 1992) and fires and extractive uses in Colorado (Price 1991). These changes in vegetation and land use are often not climate related (Dale 1997) and are not included in GCM depictions of the earth's land surface. Even when climate scenarios are used to drive ecological or economic models, the climate-related changes in land cover and use projected in the ecological and economic models do not feed back to the climate models in most cases.

Some investigators have shown the impact of land use on regional climates. Pielke et al. (1997) used land use data to demonstrate the role that landscapes (particularly spatial heterogeneity) have on the development of weather disturbances, such as thunderstorms in the Great Plains. The urban heat island effect, where large masses of concrete absorb solar radiation, is well-documented. Bonan (1997) used a simulation model to examine the impact of the cumulative changes in land cover and land use in the United States on climate in the United States. Modern vegetation includes crops replacing grassland vegetation in the central U.S. and the needleleaf evergreen, broadleaf deciduous, and mixed forests of the eastern U.S. The modeling exercise indicated that temperatures were 1 degree C cooler in the eastern U.S. and 1 degree C warmer over the western U.S. in spring. Bonan (1997) reported that the sulfate aerosols in the atmosphere in the eastern U.S. offset the warming impact of the greenhouse gases locally there. A clearer understanding of how land use affects local climate will be important in managing landscapes under an altering climate.

Impact of Climate Change on Forests, Wood Products, and Carbon

Forest Service RPA and Global Change Research Program Assessment of Climate Change

To develop forest policy actions to meet the challenges and opportunities of climate change, an integrated assessment is needed where climate information, forest productivity, forest management, the demand for forest products, and carbon sequestration is considered holistically. As described in Watson et al. (1998), current approaches to integrated assessments fall into three main categories: 1) the "vertical" dimension, where integration occurs through the chain of effects from changes in atmospheric composition and climate to changes in biophysical systems to socioeconomic consequences; 2) the "horizontal" dimension, which emphasizes the interactions among systems, sectors, and activities; and 3) the "time" dimension, where trends in society are projected over the transient path of the projected climate. Each of these approaches offers important insight into questions surrounding the impact of climate and climate variability on the forest environment and economy. The most recent RPA climate change assessment was based on the vertical approach, with some consideration of the temporal dynamics (fig. 1.2). This report, in cooperation with the Forest Service Global Change Research Program, seeks to establish the foundation for the next quantitative analyses of the impact of climate change on forests.

The Forest Service Global Change Research Program (FSGCRP) was initiated in the late 1980s to provide the scientific basis to address three broad questions (Birdsey et al. 1997): 1) What processes in forest ecosystems are sensitive to physical and chemical changes in the atmosphere? 2) How will future physical and chemical climate change influence the structure, function, and productivity of forest and related ecosystems, and to what extent will forest ecosystems change in response to atmospheric changes? and 3) What are the implications for forest management and how must forest management activities be altered to sustain forest productivity, health, and diversity? Experimental studies, monitoring, and modeling research are an integral part of the FSGCRP. Through participation in the U.S. Department of Agriculture's Global Change Research Program, the FSGCRP is a part of the U.S. Government's Global Change Research Program (USGCRP). The USGCRP has been developed under the direction of the Executive Office of the President, through the National Science and Technology Council (NSTC) and its Committee on Environment and Natural Resources (CENR).

The FSGCRP and the RPA assessments have a common goal of assessing current and future resource trends. Questions critical to understanding the impact of global climate change on current and future trends are the focus of the joint FSGCRP-RPA assessment. Six policy questions were identified (Joyce et al. 1997) and these questions form the basis for the subsequent chapters in this report.

> What are the likely effects of increasing atmospheric carbon dioxide and prospective climate changes on ecosystem productivity, as measured by changes in net primary productivity?
>
> To what geographic extent will potential ecosystem types change or move across the United States, as measured in composition and boundary changes?
>
> What changes in forest productivity will occur as measured by changes in volume, growth, and biomass?
>
> What are the potential impacts on the forest sector under climate change, as measured by employment and timber prices?
>
> When forest policy questions for the RPA Assessment, such as reduced NFS harvest, are examined with and without climate change, do the forest sector impacts differ greatly in magnitude or kind?
>
> What are the opportunities and costs of emissions mitigation using forest ecosystem management and forest products technologies?

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CHAPTER 2

Biome Redistribution Under Climate Change

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Introduction/Background

General warming in the Northern Hemisphere has been recorded since the end of the 1800s following the Little Ice Age (Folland et al. 1990). Records of glacier retreat during the last 100 years over the entire globe (Oerlemans 1994) independently confirmed the recorded trend in global temperature rise. Several studies have illustrated various responses to this climate forcing, i.e., the recorded changes in temperature and precipitation concurrent with the increase in atmospheric CO_2 concentration, increases in density of tree populations (Morin and Payette 1984; Payette and Filion 1985; Scott et al. 1987), declines in tree populations (Hamburg and Cogbill 1988), treeline displacement (Lescop-Sinclair and Payette 1995) or lack thereof (MacDonald et al. 1998), lengthening of the growing season (Mynemi et al. 1997), and enhanced tree growth (Jacoby et al. 1996). It is critical that we identify the tools needed to estimate potential consequences of climate change on forest ecosystems (Joyce and Birdsey this volume) and develop management practices and policies adapted to projected drifts in the geographic distribution of ecosystems.

Emanuel et al. (1985), who used the Holdridge lifezone model (Holdridge 1947), and Box (1981) were among the first to use correlational models between average climate and vegetation distribution to predict the responses of vegetation to climate change using general circulation model (GCM) climate simulations. The Holdridge lifezone classification relates the distribution of major ecosystems to mean annual biotemperature, mean annual precipitation, and the ratio of potential evapotranspiration to precipitation (Holdridge 1947). It was used by several authors (Emanuel et al. 1985; Prentice and Fung 1990; Smith et al. 1992) to examine potential global shifts in major ecosystems with climate change (Dale 1997). Results from Smith et al. (1992) showed a global decrease in the extent of tundra and desert, with a concurrent increase in grassland area, under four different GCM climate change scenarios. Results also showed an increase in tropical forest area and the replacement of tundra by boreal forests. These static models offer simplicity and availability but: 1) they do not take into account seasonality; 2) they have strict climate boundaries which create problems for representing transitional vegetation; and 3) they cannot include any direct CO_2 effect or indicate changes in vegetation density, runoff, or nutrient fluxes.

Over 100 "gap" model studies have also been conducted to simulate the impacts of global change on forests (Smith and Shugart 1996; Dale and Rauscher 1994; Smith et al. 1992). These models predict the establishment, growth, and death of individual trees for all potential species on a site. They include a wide range of disturbances such as fire, blowdown, insect defoliation, and drought. Simple rules are used to simulate succession in most forests. Comparative studies showed that seemingly similar models could yield totally different projections of future forest composition (for example, Bugmann 1997), since there is considerable uncertainty about the appropriate formulation of environmental influences on demographic processes. Early versions of gap models had been developed for current climate. Their applicability to changing climate conditions and increasing CO_2 concentration was questionable (for example, Loehle and Leblanc 1996). However, a second generation of gap models was developed with improved formulations of key relationships, including physiological mechanisms, thus allowing more mechanistic calculations of environmental effects on tree growth. Functional types were used to reduce the numbers of site-specific parameters required to run the models (Friend et al. 1997). Unfortunately, there has not been enough time yet for results from climate change research with these newer models to be widely circulated and published.

Biogeography models such as DOLY (Woodward and Smith 1994), MAPSS (Neilson 1995), and BIOME2 or 3 (Haxeltine et al. 1996), which are based on ecophysiological constraints and resource limitations, have been considered the next generation of equilibrium spatial models (Monserud and Leemans 1992). They are capable of simulating impacts on natural vegetation at all scales from global to continental, regional, and local (Smith et al. 1994) and have been used in several global climate change studies (IPCC 1996; VEMAP Members 1995; Neilson et al. 1998).

The objective of this chapter is to address the following question: To what geographic extent will potential ecosystem types change or move across the United States, as measured in composition and boundary changes? To do so, we used results from three different studies (Neilson et al. 1998; VEMAP Members 1995; Neilson and Drapek 1998), which are summarized in table 2.1. Three different models (DOLY, MAPSS, BIOME2 and its later version BIOME3) were run at two spatial resolutions (half-degree latitude × half-degree longitude, and 10 km) for two geographic extents (North America and the conterminous United States). Older and newer GCM-generated climate scenarios were used to describe the impacts of

Table 2.1—Summarized description of the three studies used in this article to illustrate the impact of climate change on biome distribution. FAR = First Assessment Report (IPCC 1990) including climate change scenarios from GFDL-R30, GISS, OSU, UKMO; SAR = Second Assessment Report (IPCC 1996) including climate change scenarios from HADCM2SUL and HADCM2GHG (see table 3 for details on scenarios).

Region of study (project)	Biogeography models	Resolution	Reference	Climate change scenario	Climate data source
North America	MAPSS BIOME3	0.5° latitude × longitude	Neilson et al. 1998	FAR and SAR	Leemans and Cramer 1991
Conterminous USA (VEMAP)	MAPSS DOLY BIOME2	0.5° latitude × longitude	VEMAP Members 1995	FAR	Kittel et al. 1995
Regional USA	MAPSS	10 km	Neilson and Drapek 1998; Borchers and Neilson 1998	FAR and SAR	NOAA-EPA 1997

the improvements made in projecting future climates on ecological simulation results. The rationale for using this approach is that: 1) focusing on the entire North American continent enables us to include entire biomes regardless of political boundaries; 2) focusing on the U.S. enables us to address nationally relevant issues and to compare MAPSS results with other model projections; and 3) a 10 km resolution is a more adequate scale to focus on regional impacts. A different baseline climatic dataset was used for each of the three studies, which explains the differences between the North American study and VEMAP, both of which were performed at the same half-degree resolution. Using results from these studies increases the information gain about U.S. forests and also emphasizes the uncertainties associated with the results.

Methodology

Biogeography Models

Models

Process-based biogeography models simulate the dominance of various plant lifeforms in different environments based on ecophysiological constraints, such as growing degree days and minimum winter temperatures, and resource limitations such as available soil water for plant uptake and available sunlight for the understory canopy (VEMAP Members 1995). These models simulate potential "climax" vegetation at steady state under any climate, past, present, or future (Neilson and Running 1996).

Most of the results presented in this chapter come from the MAPSS (Mapped Atmosphere Plant Soil System) model (Neilson 1995; Neilson and Marks 1994). It includes a water submodel that calculates plant available water and a rule-based submodel that determines the climatic zone, the lifeform, and the plant type as a function of temperature thresholds and water availability. The maximum potential leaf area index (LAI) a site can support is calculated iteratively. It uses an aerodynamic approach sensitive to canopy characteristics to calculate evapotranspiration. Grasses and trees have different rooting depths in a multi-layer soil and compete for available soil water, while shading by trees limits grass growth. Vegetation classification in MAPSS is based on the presence/absence and LAI values of three types of lifeforms—trees, shrubs, and grasses—with their leaf characteristics, thermal affinities, and seasonal phenology. The woody components, trees or shrubs, are assumed to be dominant and mutually exclusive. MAPSS includes a fire submodel that maintains transition zones such as the prairie peninsula. The model has been run at two different resolutions: 1) 10 km; and 2) half degree latitude-longitude resolution for VEMAP and the North American study.

BIOME2 and DOLY are two other biogeography models that have been compared to MAPSS in VEMAP (VEMAP Members 1995). The newer version of BIOME2, BIOME3, was later compared to MAPSS in the North American study (Neilson et al. 1998). BIOME3 builds upon BIOME2 but contains a more process-based canopy physiology, optimizing carbon gain through photosynthesis with radiation and water balance constraints on stomatal conductance. In BIOME2 and BIOME3 (Haxeltine and Prentice 1996; Haxeltine et al. 1996; Prentice et al. 1992), plant functional types (PFT) are calculated using a small set of ecophysiological constraints such as minimum temperature tolerance. Gross primary production (GPP) is calculated for each PFT as a function of photosynthetically active radiation (PAR) based on the Farquhar photosynthesis equation (Farquhar et al. 1980). GPP is then reduced by soil water availability and temperature limitations. Foliar projected cover (or leaf area index, LAI, in the case of BIOME3) is calculated to maximize net primary production (NPP). Evapotranspiration is determined by available energy. Grass and woody vegetation compete for water as a function of their rooting depth in a hydrology submodel. Fire and light competition are empirically simulated in the model.

DOLY (Woodward and Smith 1994; Woodward et al. 1995) simulates photosynthesis using the Farquhar photosynthesis equation (Farquhar et al. 1980) and evapotranspiration using the Penman-Monteith equation (Monteith 1981). NPP is affected by temperature and nitrogen availability. N uptake is a function of soil carbon and nitrogen contents, temperature, and moisture. Maximum leaf area is constrained by radiation, water balance, and nitrogen.

DOLY, BIOME2, and MAPSS all incorporate some sort of direct response to changes in CO_2 concentration, but they differ in the specific mechanisms considered. In MAPSS, stomatal conductance is reduced by elevated CO_2 concentration, which leads to a reduction in evapotranspiration and—indirectly—increased LAI. The model, however, does not allow for any direct CO_2 effect on the competitive balance between C3 and C4 grasses. In BIOME2, the impact of CO_2 concentration is included in the photosynthesis algorithm where it can affect C3 vs. C4 competition, but does not directly affect water balance. DOLY includes CO_2 concentration in the calculation of photosynthesis and evapotranspiration, but does not include a direct CO_2 effect on the competitive balance between C3 and C4 grasses. Additional information on the models can be found in VEMAP Members (1995) where detailed comparison tables summarize their differences.

The models require latitude (BIOME), mean monthly or daily (DOLY model only) temperature, precipitation, humidity (DOLY and MAPSS), wind speed (DOLY and MAPSS), and solar radiation (BIOME and DOLY). MAPSS and BIOME3 were driven in the North American study by a baseline long term average monthly climate dataset, which corresponds to an improved version of that described in Leemans and Cramer (1991), and was obtained from the Cramer and Leemans database (W. Cramer, personal communication). In VEMAP, the three biogeography models (MAPSS, BIOME2, and DOLY) used a baseline dataset that was interpolated from a large number of U.S. weather stations (4613) and is described in Kittel et al. (1995). The method used to create the baseline climate dataset used by MAPSS at the 10 km-resolution is described in detail in Borchers and Neilson (1998). The dataset includes information between 1211 and 4613 stations (depending on the variable calculated) from the conterminous United States (NOAA/NGDC 1997).

The models also require soil texture (sand, silt, clay fraction) and soil characteristics such as depth and rock fragment content. U.S. soils data are based on the 10 km gridded National Soil Geographic (NATSGO) data base modified by Kern (1994, 1995). For the VEMAP project, cluster analysis grouped the 10 km subgrid elements into one to four dominant soil types for each half degree cell. Cell soil properties were then represented by one or more dominant soil profiles rather than by the average one that may not correspond to an actual soil in that region. For the runs over North America, the digital version of the FAO soils map of the world was used.

Vegetation Types

MAPSS includes 45 vegetation types. In VEMAP, Küchler's (1964) map of potential vegetation was aggregated to 22 classes. To simplify result analysis in this chapter, we used a simplified classification aggregated into 10 vegetation categories. Table A1 in Appendix A illustrates the correspondence between our simplified categories, the MAPSS vegetation types, and the VEMAP classes. Table 2.2 summarizes the thresholds that MAPSS uses to distinguish the vegetation categories.

Tundra and Taiga-Tundra exist beyond the climatic limit of the boreal forest. Beyond treeline, the cold-dominated landscapes with permanently frozen soils are characterized by tundra vegetation composed of shrubs, grasses, mosses, and lichens. Cryptogams are abundant. The boreal forest-tundra ecotone corresponds to the taiga-tundra zone, which can extend up to 235 km in width in central Canada and 300 km in Quebec. It corresponds to the limit beyond which the forest tree life cycle is interrupted and sexual regeneration is either irregular or unsuccessful (Lenihan and Neilson 1993). This vegetation category in MAPSS also includes the high-altitude alpine ecosystem located mostly in the western third of the United States. It is sometimes called "alpine tundra" because migrations of arctic plants during Pleistocene and Holocene resulted in alpine floras with a strong arctic component. However, the alpine flora can be much more diverse and richer than the arctic flora. Only low growing season temperatures are in common with the arctic tundra, and much variation exists in their physical environments (solar radiation, daylength, soil, snowpack, topography). It is located above the upper limits of forests and consists mostly of dwarf shrubs and short perennial herbaceous plants.

The boreal coniferous forest is constrained by cold temperatures to the north which limit forest stature and reproduction. The southern limits of the boreal coniferous forest are generally defined by their juxtaposition with temperate forests or with interior savanna woodlands and grasslands. Boreal tree species can generally grow further south but are outcompeted by temperate hardwoods and conifers, which are limited by cold temperatures from spreading further north (Lenihan and Neilson 1993; Starfield and Chapin 1996). As with tundra and taiga-tundra, there is a mid-latitude equivalent of

Table 2.2—Summary of the rules used to define the simplified vegetation classes in MAPSS. Under cold conditions, energy constraints are represented by the sum of growing degree days. For milder conditions, minimum monthly temperatures (or their equivalent), LAI, duration of the growing season, and associated available water are used to classify the vegetation. A monthly mean temperature of –16°C is approximately equivalent to an absolute minimum temperature of –40°C, the supercooled freezing point at water which limits the northward spread of most temperate deciduous trees. A monthly mean temperature of 13°C separates subtropical regions, where some frost occurs during the year, from tropical regions where no frost occurs during the year. Forests are assumed to have an LAI value greater than 3.75. Dry summers (mean summer precipitation below 40mm) characterize temperate evergreen forests. When there is enough rainfall in the summer but the growing season is too short, the vegetation is classified as short temperate mixed forests. We use the ratio of AT over LAI as an index of growing season productivity required to meet year-long respiration demand, to characterize the growing season length. GDD = growing degree days (base 0°C); MMT = monthly mean temperature; LAI = leaf area index with LAIg for grass LAI, LAIs for shrub LAI, and LAIt for tree LAI; MSR = minimum summer rainfall; AT = actual transpiration.

Vegetation classes	GDD (°C)	MMT (°C)	LAI	MSR (mm)	AT/ LAI
Tundra	< 735*				
Taiga - Tundra	735* < GDD < 1330*				
Boreal coniferous forest	> 1330*	< –16	LAI_t => 3.75		
Temperate evergreen forest		–16 =< MMT < 14	LAI_t => 3.75	< 40	
Temperate mixed forest		–16 =< MMT < 14	LAI_t => 3.75	> 40	< 10
Tropical broadleaf forest		> 14	LAI_t => 3.75		
Savanna woodland			2 < LAI_t < 3.75		
Shrub woodland			LAI_s > 0.7		
Grasslands			LAI_g > 0.1		
Arid lands			LAI_g < 0.1		

*For alpine rather than boreal environments, GDD thresholds used are 615 and 1165°C.

this vegetation type in high mountainous terrain. In the North American study, taiga, tundra and boreal forest vegetation types are mostly referred to as arctic types. In the VEMAP and 10 km-resolution study, the three types only correspond to their mid-latitudinal definitions.

Temperate evergreen forests, such as in the northeast U.S., tend to occur in areas that are warm enough for photosynthesis during the cool parts of the year, but that are often too cold for deciduous species to fix sufficient carbon during the frost-free season (Woodward 1987). Areas with dry summers, such as the Pacific Northwest, also tend to favor conifers or hardwoods with water conserving leaves (Waring and Schlesinger 1985; Neilson 1995). Summer drought and winter chilling required for seed set and to confer frost hardiness are critical climate factors rendering these forests sensitive to global warming (Franklin et al. 1991). Temperate mixed forests (mixed hardwood and conifer) are bound by cold temperatures to the north and the subtropical dry regions to the south (Caribbean coast in North America). They tend to occur in areas that are wet all year. The southeastern U.S. pines within this type are among the most important commercial species on the continent. Tropical broadleaf forests are currently confined to the subtropical regions of Central America.

Grasslands are the largest of the natural biomes in the United States, covering more than 125 million ha (Barbour and Billings 1988). Most of the productive arable lands in North America are former grasslands. They include the tall-grass, mixed-grass, and short-grass prairies of the central plains, the desert grasslands of the Southwest, the California grassland and the Palouse prairie in the Intermountain West. Their climates have distinct wet and dry seasons and are noted for temperature and precipitation extremes. Periodic drought and fire are important processes for limiting woody encroachment into grasslands. Arid lands, or deserts, have warm to cool climates with low rainfall and high rates of evaporation. North American deserts are often thought of as semi-desert because of their relatively lush vegetation. Savanna woodland is a broad class ranging from almost closed-canopy woodlands to very open grasslands with occasional trees. It includes pinyon-juniper woodlands, oak scrub, and the prairie peninsula region in Illinois and Indiana. Shrub woodland includes the sagebrush steppe of the Intermountain West and the Southwestern chaparral and mesquite woodlands.

Climate Change Scenarios

FAR Scenarios

Most published climate change impact studies have been based on equilibrium experiments from a set of

Atmospheric General Circulation Models (GCM) with simple mixed-layer oceans and prescribed land-surface properties that were run with doubled CO_2 radiative forcing. Doubled CO_2 radiative forcing (2 x CO_2) includes about 50 percent actual CO_2 forcing, and other greenhouse gases account for the remainder. At the time of the First Assessment Report (FAR) of the IPCC (Intergovernmental Panel on Climate Change) (IPCC 1990), scenarios of future climate were produced by running those GCMs to equilibrium and producing an average climate for both current and doubled CO_2 conditions (Cubasch and Cess 1990). Simulations presented in this paper rely upon such scenarios generated by the UKMO (Mitchell and Warrilow 1987), GFDL-R30 (IPCC 1990), GISS (Hansen et al. 1988), and OSU (Schlesinger and Zhao 1989) models. FAR climate scenarios were supplied by the Data Support Section within the Scientific Computing Division of the National Center for Atmospheric Research (NCAR).

SAR Scenarios

Recent climate change impact studies have been based on GCM transient CO_2 experiments with coupled atmosphere and ocean. By the Second Assessment Report (SAR) of the IPCC (Gates et al. 1996), the atmospheric-oceanic GCMs were simulating time series of climatic changes and some included sulfate aerosols that could regionally cool the climate. Some of the analyses presented in this chapter relied on such simulations from the Hadley Centre (Johns et al. 1997; Mitchell et al. 1995; IPCC 1996): HADCM2GHG scenario and HADCM2SUL scenario with sulfate aerosols. These scenarios were extracted from transient GCM simulations in which trace gases were allowed to increase gradually over a long period of years, allowing the climate to adjust while incorporating inherent lags in the ocean-atmosphere systems.

Spatial and Temporal Resolution of the Scenarios

The coarse grid of the GCM scenarios was interpolated to a half degree latitude-longitude resolution or a 10 km Albers grid using a 4 point inverse distance squared algorithm in a raster-based Geographic Information System (U.S.A.-CERL 1993). Ratios ($(2 \times CO_2) / (1 \times CO_2)$) were applied to all climate variables (except temperature) from a baseline long-term average monthly climate dataset (Leemans and Cramer 1991). Ratios were used to avoid negative numbers, but were not allowed to exceed 5 to prevent unrealistic changes in regions with normally low rainfall. Temperatures were calculated as a difference $(2 \times CO_2) - (1 \times CO_2)$ and applied to the baseline climate dataset.

The control climate ($1 \times CO_2$) was extracted from transient GCM simulations as a 30-year model output average associated with present climate (1961–1990). The future climate ($2 \times CO_2$) scenario was extracted as a 30-year average from the time period approximating $2 \times CO_2$ forcing (2070–2099) when simulations had only attained about 64 percent to 68 percent of the eventual equilibrium temperature change, due to thermal lags in the oceans. Therefore, the two Hadley scenarios discussed here produced relatively modest warming compared to other SAR scenarios.

Description of the Scenarios

FAR climate scenarios are described in detail in Cubasch and Cess (1990). Over the conterminous United States, the OSU scenario is the coolest with small increases in precipitation, the GISS is warm and relatively dry, the GFDL-R30 is warm and extremely wet, and the UKMO scenario is very warm and moderately wet (table 2.3). Over the land areas of the world, the OSU scenario is still the coolest but quite wet. Both GISS and GFDL-R30 are warm and wet, and UKMO is hot and drier than the three other scenarios.

The SAR climate scenarios from Hadley Centre are in general cooler scenarios (table 2.3) both over the North American land areas (Kattenberg et al. 1996) and over the United States. The sulfate aerosol scenario is very dry over the world but quite wet over the United States. Addi-

Table 2.3—Simulated changes in temperature (°C) and precipitation (percent) over the world land area and over the conterminous U.S. From the First Assessment Report (FAR) of the Intergovernmental Panel on Climate Change (IPCC), the atmospheric general circulation models (GCMs) used to simulate climate were: the Oregon State University model (OSU), the Goddard Institute of Space Studies (GISS) model, the Geophysical Fluid Dynamics Laboratory model (GFDL-R30), and the United Kingdom Meteorological Office model (UKMO). From the Second Assessment Report (SAR) of the IPCC, the atmospheric-oceanic GCM used to simulate climate was the Hadley Centre Transient general circulation model with (HADCM2SUL) and without (HADCM2GHG) sulfate aerosol forcing.

	Temperature (°C)		Precipitation (%)	
	World	U.S.	World	U.S.
FAR scenarios				
OSU	3.0	3.0	20.9	2.1
GISS	4.3	4.4	16.0	5.1
GFDL-R30	3.9	4.2	18.7	18.9
UKMO	6.0	6.6	15.1	11.3
SAR scenarios				
HADCM2GHG	4.3	3.7	13.2	30.7
HADCM2SUL	3.5	2.8	1.7	22.9

tional descriptions and comparisons of FAR and SAR scenarios can be found in Neilson and Drapek (1998).

Results

North American Impacts

Effects of CO_2: MAPSS Results With FAR Scenarios

Results from running the equilibrium vegetation distribution model MAPSS for the North American region (half degree latitude-longitude resolution) with and without a CO_2 effect for three GCM scenarios are illustrated in table 2.4. Major agreements can be identified: 1) decreases in the area of both tundra and taiga-tundra (from 42 to 70 percent and from 39 to 56 percent respectively); 2) increases in the areal extent of savannas (from 14 percent with the CO_2 effect to 125 percent without); and 3) decreases in the area of shrublands (from 2 to 20 percent). When the CO_2 effect is included (35 percent decrease in stomatal conductance), temperate evergreen and temperate mixed forests are predicted to increase in area, with the largest increases predicted under the UKMO scenario (80 percent and 41 percent respectively).

Comparison Between BIOME3 and MAPSS Using SAR Scenarios

MAPSS simulations using the Hadley Centre scenarios (HADCM2SUL and GHG) are compared to those of BIOME3 in table 2.5. The models predict opposite trends for both the boreal coniferous forest and the temperate evergreen forest. BIOME3 groups boreal forest and taiga-tundra into one vegetation type while MAPSS does not. MAPSS predicts large decreases in the taiga-tundra area (eastern Canada and Alaska, fig. 2.1), which matches BIOME3 simulations of boreal forest area increase, but MAPSS also simulates small increases in the boreal forest area proper. Thus, the two models are consistent with each other with respect to high-latitude ecosystems, with apparent differences only indicating different classification schemes. For all scenarios, both models agree on simulating: 1) large decreases in the area of pure tundra replaced by the warmer taiga-tundra; 2) large increases in the temperate mixed forest (table 2.5) moving westward in the United States and northward into Canada (fig. 2.1); and 3) large decreases in the area of arid lands (table 2.5) that are replaced by grasslands (fig. 2.1). When the CO_2 effect is not included in the models, both simulate increases in the extent of savannas and grasslands.

With the HADCM2SUL scenario, MAPSS simulates large shifts of Northwest temperate evergreen forests to Alaska replacing the taiga-tundra area (fig. 2.1) but BIOME3 shows a much smaller expansion (not shown here, fig. C-4 and 5 in Neilson et al. 1998). They both simulate an expansion of the southeastern temperate mixed forest at its western edge.

Conterminous U.S. Impacts

Comparison Between DOLY, BIOME2, and MAPSS Using FAR Scenarios

Three biogeographical models (DOLY, BIOME2, and MAPSS) were used to simulate vegetation distribution for current climate conditions and future climate conditions under three different GCMs. The three biogeography models produce similar maps of current vegetation distribution. Whether the CO_2 effect is included or not, MAPSS and BIOME2 simulate a loss of alpine tundra and boreal coniferous forest area (table 2.6). DOLY simulates an increase in alpine tundra when the CO_2 effect is included for all climate change scenarios. DOLY only simulates an increase in the extent of the boreal coniferous forest under the OSU scenario when the CO_2 effect is included. When the CO_2 effect is not included, MAPSS simulates a decrease in temperate forests accompanied by an increase in savannas and grasslands; on the other hand, BIOME2 simulates an increase in temperate forests and tropical broadleaf forest areas at the expense of savannas, shrublands, and arid lands. Both DOLY and MAPSS simulate increases in arid land area. DOLY produces a greater expansion of forests into the Great Plains and produces little forest dieback under altered climate (not shown here). However, DOLY also produces far more dramatic increases in the extent of Southwest deserts than either MAPSS or BIOME2 (table 2.6). The only general agreements in VEMAP for all scenarios and all models are that shrubland area decreases when the effect of CO_2 is included, and when it is not, tundra and boreal forest areas decrease. A more detailed analysis of these results is presented in Neilson and Chaney (1997).

Effects of CO_2: MAPSS Results with FAR Scenarios

There are large differences between MAPSS results whether the effect of CO_2 is included or not. For example, MAPSS simulated large decreases (40–80 percent) in the area of temperate mixed forest in the eastern United States for "warmer" scenarios such as the UKMO and GFDL when the CO_2 effect was not included. These decreases were greatly reduced (12–14 percent) when water use effi-

Table 2.4—Percentage of simulated area for each simplified vegetation type under current climate for North America at a half degree latitude-longitude resolution by the biogeography model MAPSS. Total area is 18,923 10^3 km^2. Percentage change in area for each vegetation type from current climate to future climate conditions with no CO_2 effect (A) and with CO_2 effect (B). Percentage change in vegetation type area is calculated as: (scenario – current)/current. The atmospheric general circulation models used to simulate climate (FAR scenarios) were: the Oregon State University model (OSU), the Geophysical Fluid Dynamics Laboratory model (GFDL-R30), and the United Kingdom Meteorological Office model (UKMO).

	Current (% of total land area)	OSU (Δ %)	GFDL-R30 (Δ %)	UKMO (Δ %)
A. With no CO_2 effect:				
Tundra	16	–42	–58	–70
Taiga–Tundra	16	–39	–39	–56
Boreal coniferous forest	16	+18	+40	–15
Temperate evergreen forest	7	–3	–6	+8
Temperate mixed forest	16	–18	+49	–28
Savanna woodland	12	+71	+104	+125
Shrub woodland	7	–4	–2	–10
Grassland	9	+36	+59	+114
Arid land	2	+100	–3	+113
B. With CO_2 effect:				
Tundra	16	–42	–58	–70
Taiga–Tundra	16	–39	–39	–56
Boreal coniferous forest	16	+35	+50	–13
Temperate evergreen forest	7	+34	+30	+80
Temperate mixed forest	16	+29	+7	+41
Savanna woodland	12	+14	+44	+52
Shrub woodland	7	–15	–20	–2
Grassland	9	–<1	+23	+45
Arid land	2	+5	–64	+17

ciency was increased (table 2.6). With a milder scenario like OSU, these forests could even increase in their extent by about 10 percent if stomatal conductance is reduced up to 35 percent by elevated CO_2 concentration as it is assumed in MAPSS (table 2.6). In the early stages of warming, when temperature increases are small, a CO_2-induced increase in water use efficiency could result in an expansion of temperate forests into neighboring drier areas, and a concurrent increase in forest density throughout much of the current forest distribution. However, as the CO_2 effects saturate and temperatures continue to increase, the elevated evaporative demand could then overwhelm the increased water use efficiency, and temperate forests could contract in area and undergo a drought-induced decline in vegetation density (Neilson and Drapek 1998). Complex responses of the vegetation to changes in their climatic environment and in the atmospheric CO_2 concentration are to be expected. Early responses to the CO_2 fertilization effect leading to a greening of the land may be followed by forest diebacks due to increased warming and drought stress.

Regional Impacts

Results From MAPSS and Other Models Using FAR and SAR Scenarios

MAPSS simulations at the 10 km resolution using the Hadley Centre sulfate aerosol scenario (fig. 2.1 and table 2.7) show that coniferous forests in northern Minnesota, Wisconsin, and Michigan (upper peninsula) are displaced by temperate mixed forests expanding from the east and south. For all FAR scenarios, MAPSS simulates large decreases in the temperate mixed forest and boreal forest area around the Great Lakes region, which are replaced by savannas and grasslands (figs. 2.1 and 2.2). With “warm” climate change scenarios such as the UKMO and, to a lesser extent, the GFDL-R30, MAPSS simulates the fragmentation of the southeastern temperate mixed forest, which is replaced by drier ecosystems such as savannas and grasslands (fig. 2.2). With cooler scenarios such as OSU, MAPSS simulates an increase in forested areas in and around the Willamette Valley in the Pacific Northwest and on the western edge of the southeastern forests (fig. 2.2). Mesquite-oak woodlands, currently in central Texas, would shift north into the Great Plains region while the grasslands would replace the semi-deserts of eastern Texas, southern New Mexico, western Arizona, and eastern California. Under the UKMO scenario (fig. 2.2 and table 2.7), southwestern warm-desert species could extend into cold-desert regions as far north as eastern Oregon and Washington, or in the case of the OSU scenario (fig. 2.2) remain about where they are today.

Mean annual temperatures have increased globally by 0.5°C per century, and by 1.2°C in the southwestern desert region of the United States between 1901 and 1987 (Lane et al. 1994). Emanuel et al. (1985) suggested a possible future increase of up to 17 percent in desert land area of North America. Predictions from VEMAP Members (1995) simulations (half degree latitude-longitude resolution), including both climate change and increased CO_2, show both decreases and increases in the areal extent of subtropical shrublands (southwestern deserts). Since thermal constraints have kept southwestern species from moving northward, an increase in temperature at higher latitude and sufficient available water should enable those desert species to reach the Great Basin area. In fact, MAPSS simulates a northern migration and expansion of subtropical mixed shrub savannas into the Great Basin region and as far north as eastern Washington. Yet, expansion of desert species does not necessarily imply increased desertification. In fact, MAPSS simulates up to a 200 percent increase in leaf area index (primarily grassland) in the southwestern desert region of the United States where grasses can outcompete shrubs under future wetter conditions.

Comparisons between MAPSS and the PnET model (Aber and Federer 1992) over the Southeast are consistent in terms of forest decline, but not in terms of runoff (Borchers and Neilson 1998). PnET simulated increases in annual runoff from 10 to 240 percent, as evapotranspiration was altered by climate change scenarios, and forest death was occurring without replacement (McNulty et al. 1994). PnET does not include an understory; thus when forests decline, no other vegetation types replace them and runoff increases. In MAPSS, when forests decline, shrubs and grasses increase and may use as much or more water, producing declines in runoff. PnET also simulated severe reductions in annual NPP on loblolly pine sites in Texas (–55 percent), Mississippi (–35 percent), Florida (–60 percent), and Virginia (–15 percent) with climate scenarios based on historical records from 1951 to 1984 (Aber et al. 1995; McNulty et al. 1994, 1996a and b).

Even in regions where the vegetation type would not change, it could either increase in density or decline. Where vegetation density, characterized in MAPSS by leaf area, is decreasing, some level of vegetation dieback (if forested) can be expected or at least a reduction in

Table 2.5—Percentage of simulated area for each simplified vegetation type under current climate (A) for North America at a half degree latitude-longitude resolution by the biogeography models MAPSS and BIOME3. Total area is 18,923 10^3 km^2. Percentage change in area for each vegetation type from current climate to future climate conditions with no CO_2 effect (B) and with CO_2 effect (C). Percentage change in vegetation type area was calculated as: (scenario – current)/current. The atmospheric general circulation model used to simulate climate (SAR) was the Hadley Centre model without aerosols (HADCM2GHG) and with sulfate aerosols (HADCM2SUL). BIOME3 groups together boreal forest and taiga-tundra. (+++ denotes a vegetation type that did not exist in current climate conditions. NA corresponds to a vegetation type that did not exist in either current or future climates.)

A. Current climate:	MAPSS	BIOME3
Tundra	16	17
Taiga–Tundra	16	
Boreal coniferous forest	16	33
Temperate evergreen forest	7	6
Temperate mixed forest	16	20
Tropical broadleaf forest	0	0
Savanna woodland	12	6
Shrub woodland	7	8
Grassland	9	8
Arid land	2	<0.1

	No aerosols		Sulfate aerosols	
B. Future climate with no CO_2 effect:	MAPSS	BIOME3	MAPSS	BIOME3
Tundra	–45	–47	–37	–42
Taiga–Tundra	–44	NA	–41	NA
Boreal coniferous forest	+10	–32	+12	–24
Temperate evergreen forest	+29	–18	+28	–167
Temperate mixed forest	+29	+59	+34	+42
Tropical broadleaf forest	NA	+++	NA	+++
Savanna woodland	+29	+94	+12	+84
Shrub woodland	–11	+10	+10	+14
Grassland	+46	+19	+15	+23
Arid land	–22	–100	–2	–100
C. Future climate with CO_2 effect:				
Tundra	–45	–47	–37	–42
Taiga – Tundra	–44	NA	–41	NA
Boreal coniferous forest	+16	–36	+15	–26
Temperate evergreen forest	+82	–22	+82	–15
Temperate mixed forest	+57	+98	+52	+79
Tropical broadleaf forest	+++	+++	+++	+++
Savanna woodland	+2	+20	–1	+11
Shrub woodland	–22	–<1	+1	+8
Grassland	–3	+6	–30	–5
Arid land	–78	–100	–70	–100

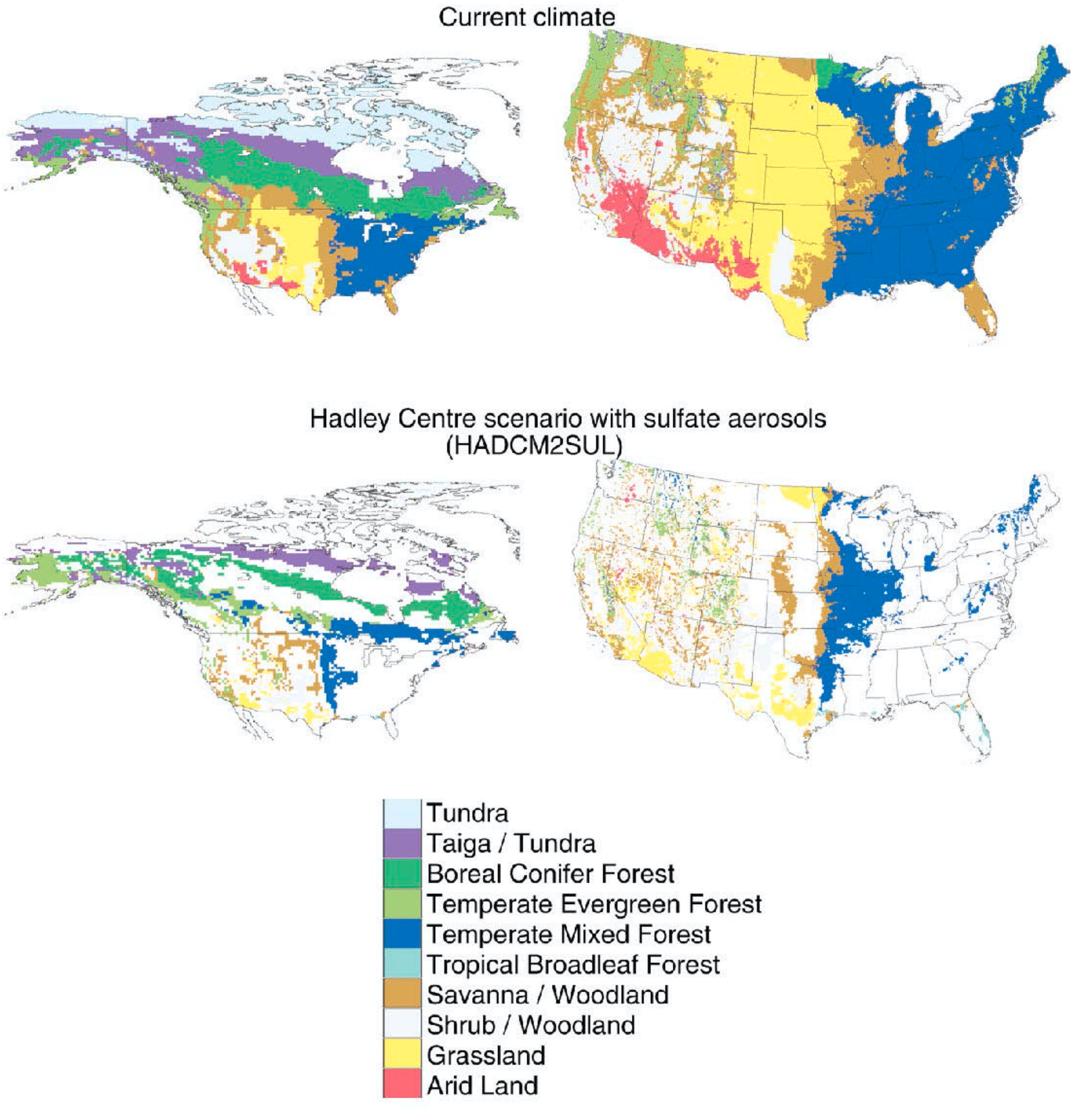

Figure 2.1–Top: Aggregated potential vegetation classes simulated for the North American region at a half degree latitude-longitude resolution and for the United States at a 10 km resolution for current climate conditions. Bottom: Areas where new vegetation classes are simulated in future climate conditions by the MAPSS model using the Hadley Centre climate change scenario including sulfate aerosols (HADCM2SUL). Areas where there is no change in vegetation type remain white.

the standing crop. Both FAR and SAR scenarios indicate that relatively large regions of the United States would undergo such reductions or increases (Neilson et al. 1998). However, "hotter," more severe scenarios, such as UKMO, indicate vegetation dieback or standing crop reduction over most of the U.S.

More Detailed Considerations About U.S. Forests With FAR and SAR Scenarios

Neilson and Chaney (1997) translated the MAPSS vegetation types into forest type categories (MAPSS assess-

Table 2.6—Percentage of simulated area for each simplified vegetation type under current climate in the conterminous U.S. at a half degree latitude-longitude resolution for the VEMAP Project (A). Three biogeography models were used: MAPSS, BIOME2, and DOLY. Total area is 7,524 10^3 km^2. Percentage change in area for each vegetation type from current climate to future climate conditions with no CO_2 effect (B) and with CO_2 effect (C). Percentage change in vegetation type area is calculated as: (scenario – current)/current. The atmospheric general circulation models used to simulate climate (FAR) were: the Geophysical Fluid Dynamics Laboratory model (GFDL-R30), the Oregon State University model (OSU), and the United Kingdom Meteorological Office model (UKMO). (+++ denotes a vegetation type that did not exist in current climate conditions. NA corresponds to a vegetation type that did not exist in current climate conditions and does not exist either in future climate conditions.)

Vegetation classes	MAPSS	BIOME2	DOLY
A. Current climate:			
Tundra	<1	1	<1
Boreal coniferous forest	2	2	3
Temperate evergreen forest	9	9	8
Temperate mixed forest	34	34	31
Tropical broadleaf forest	0	<1	0
Savanna woodland	12	12	19
Shrub woodland	12	16	17
Grassland	27	19	17
Arid land	4	5	5

	MAPSS			BIOME2			DOLY		
Vegetation classes	GFDL	OSU	UKMO	GFDL	OSU	UKMO	GFDL	OSU	UKMO
B. Future climate with no CO_2 effect:									
Tundra	–100	–100	–100	–10	–86	–100	–67	–67	–100
Boreal coniferous forest	–100	–87	–100	–81	–63	–94	–79	–33	–83
Temperate evergreen forest	–10	–49	–78	+119	+76	+24	–19	–56	–32
Temperate mixed forest	–69	–39	–82	+54	+37	+23	+16	–8	+6
Tropical broadleaf forest	NA	NA	NA	+113	+167	+533	NA	NA	NA
Savanna woodland	+97	+31	+136	–47	–58	–49	+13	–4	–20
Shrub woodland	+8	+17	–15	–79	–40	–66	–41	–60	–22
Grassland	+48	+39	+71	–25	–16	+45	–7	+62	–24
Arid land	+17	+90	+80	–83	–32	–32	+97	+167	+287
C. Future climate with CO_2 effect:									
Tundra	–100	–100	–100	–100	–86	–100	+133	+33	+67
Boreal coniferous forest	–100	–87	–100	–81	–63	–94	–54	+25	–63
Temperate evergreen forest	+142	+84	+3	+111	+66	+21	–16	–53	–30
Temperate mixed forest	–12	+9	–14	+59	+45	+40	+24	+<1	+16
Tropical broadleaf forest	NA	NA	NA	+133	+167	+533	NA	NA	NA
Savanna woodland	+15	–23	+29	–48	–57	–58	–4	–17	–30
Shrub woodland	–25	–20	–10	–77	–37	–59	–25	–40	0
Grassland	–17	–17	+21	–33	–32	+12	–8	+49	–30
Arid land	–57	+3	–40	–76	–22	–22	+28	+95	+182

Table 2.7—Percentage of simulated area for each vegetation type under current climate in the conterminous U.S. at a 10 km resolution by the biogeography model MAPSS and percentage change in area for each vegetation type from current climate to future climate conditions with no CO_2 effect (A) and with CO_2 effect (B). Total area is 7,706 10^3 km^2. The atmospheric general circulation models used to simulate climate were: the Goddard Institute for Space Studies (GISS), the Geophysical Fluid Dynamics Laboratory model (GFDL-R30), the Oregon State University model (OSU), the United Kingdom Meteorological Office model (UKMO), and the Hadley Centre model without aerosols (HADCM2GHG) and with sulfate aerosols (HADCM2SUL). Note: in B, Current (climate) is without CO_2 effect. Percentage change in vegetation type area is calculated as: (scenario – current)/current.

	Current (%)	GISS	GFDL	OSU	UKMO	CM2GHG	CM2SUL
A. No CO_2 effect:							
Tundra	<1	–92	–96	–81	–100	–96	–92
Taiga – Tundra	<1	–89	–87	–67	–96	–89	–83
Boreal coniferous forest	<1	–100	–90	–75	–100	–94	–99
Temperate evergreen forest	6	–36	–42	–33	–76	–31	–31
Temperate mixed forest	30	–42	–88	–55	–94	–5	+5
Savanna woodland	16	+31	+122	+35	+79	+9	–8
Shrub woodland	14	–9	–9	+<1	–<1	–12	+14
Grassland	28	+30	+39	+38	+56	+19	+4
Arid land	5	+90	+21	+72	+118	–3	+1
B. With CO_2 effect:							
Tundra	<1	–92	–96	–81	–100	–96	–92
Taiga – Tundra	<1	–89	–87	–67	–96	–89	–83
Boreal coniferous forest	<1	–100	–90	–73	–100	–94	–99
Temperate evergreen forest	6	+9	+49	+14	–43	+23	+23
Temperate mixed forest	30	+1	–36	–15	–45	+26	+28
Savanna woodland	16	+11	+77	+16	+58	+2	–2
Shrub woodland	14	–8	–20	–8	+2	–26	+2
Grassland	28	–5	+5	+7	+25	–4	–18
Arid land	5	+0	–47	+1	+19	–69	–66

ment classes) corresponding to an aggregation of the RPA forest types. We use the same approach (table A2 in Appendix A) to present some of our results with more details about specific forest types.

With SAR scenarios, GISS, and GFDL-R30, MAPSS (at the 10 km resolution) simulates an overall increase in total forest area and small decreases (1 to 12 percent) with OSU and UKMO scenarios (table 2.8). Using either FAR or SAR scenarios, the model simulates decreases (14 to 100 percent) in northeast mixed forest area, especially mixed woodlands (100 percent), which are replaced by southeast mixed pines and hardwood forest type that are moving northward and expanding in area (by 25 to 57 percent, table 2.8). With FAR scenarios, MAPSS simulates a decrease in eastern hardwood forests (11 to 99 percent); with SAR scenarios, it simulates an increase except in the case of the oak-hickory forests, which are predicted to decrease by 16 percent when the HADCM2SUL scenario is used.

With both FAR and SAR scenarios, MAPSS simulates a large decrease in the area of western fir and spruce forests (71 to 98 percent), an increase in coastal spruce, hemlock, and redwood forests (11 to 503 percent), and an increase in western hardwoods (6 to 681 percent). With SAR scenarios, the model simulates increases in the area of other types of western forests while with FAR scenarios results are less clear. With all scenarios, the model simulates increases in tropical forest areas replacing the southeast mixed pines and hardwood forest. MAPSS also simulates decreases in arid woodland areas and increases in Mediterranean shrublands with SAR scenarios and GISS.

Implications of Biome Redistribution Results on Ecosystem Processes

Carbon Pools: Sources and Sinks

Climate change affects temperature- and moisture-controlled processes such as production and litter decomposition. Lifeform changes due to shifts in climate also affect carbon inputs. An important impact of future climate change is the projected reduction of tundra and taiga ecosystems, which may be reduced by as much as 40 to 50 percent of their present size in North America (table 2.5). The impact on the regional storage of carbon in the higher latitudes of North America may result in a shift from a net

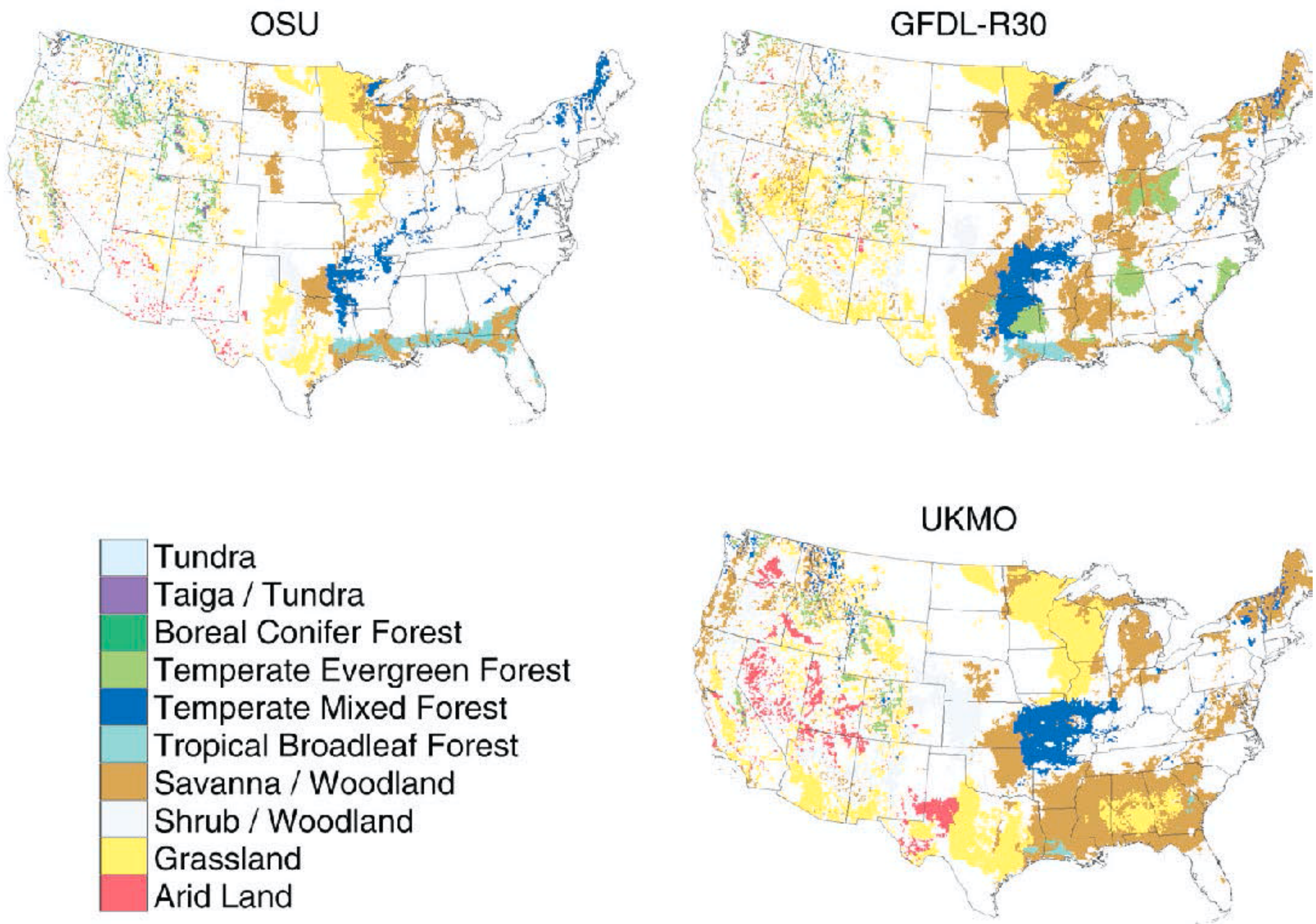

Figure 2.2—Areas where new vegetation classes are simulated in future climate conditions by MAPSS using 3 FAR climate change scenarios: OSU, GFDL-R30, and UKMO, in the conterminous United States at a 10 km resolution. Areas where there is no change in vegetation type remain white.

sink (sequestration of carbon) to a net source (release of carbon) of CO_2 (Anderson 1991; Oechel et al. 1993). Soil warming would also affect methane fluxes from tundra plant communities directly affected by drier soil surfaces and the resulting increased surface oxidation. The frozen soils of boreal forests contain one of the largest pools of carbon (Dixon et al. 1994; Gorham 1991) in the terrestrial biosphere: 200–500 Gt of carbon (1Gt = 10^9 metric tons). Goulden et al. (1998) used eddy correlation, chamber, and laboratory techniques to measure carbon balance in a typical black-spruce boreal forest site in Canada. They concluded that the deep soil carbon pool was not in equilibrium and discussed the possibility that soil C losses might be due to climate warming since Oechel et al. (1993) already reported such findings. Projected shifts in vegetation types due to climate warming would probably accentuate soil carbon losses.

Also see Heath and Smith (this volume), Smith and Heath (this volume), Birdsey (this volume), and Skog and Nicholson (this volume) for additional discussions of carbon sequestration in forests and wood products.

CO_2 Impacts on Physiological Processes

Elevated CO_2 has been documented to increase productivity, nitrogen efficiency, and water-use efficiency (IPCC 1996; Bazzaz et al. 1996). Wullschleger et al. (1995) reviewed 58 studies where a doubling of atmospheric CO_2 concentration resulted in a 32 percent average increase in plant dry mass. Norby (1996) studied seven broadleaf species under a doubling of atmospheric CO_2 concentration over a wide range of conditions, and recorded a 29 percent increase in annual growth per unit leaf area. Eamus (1991) reported reductions of leaf conductance to

Table 2.8—Forest area (in 1,000 km[2]) as described by assessment classes (Neilson and Chaney 1997) under current and future climate conditions. The atmospheric general circulation models used to simulate climate were: the Goddard Institute for Space Studies (GISS), the Geophysical Fluid Dynamics Laboratory model (GFDL-R30), the Oregon State University model (OSU), the United Kingdom Meteorological Office model (UKMO), and the Hadley Centre model without aerosols (HADCM2GHG) and with sulfate aerosols (HADCM2SUL). "Current" corresponds to current climate conditions.

Assessment classes	Current	GISS	GFDL-R30	OSU	UKMO	CM2GHG	CM2SUL
NE mixed conifers and hardwoods	536	181	18	99	29	302	462
NE mixed woodlands	52	0	0	0	0	0	0
Maple-beech-birch	210	184	18	187	3	488	723
Oak hickory forest	602	467	238	333	51	756	506
SE mixed pines and hardwoods	946	1487	1184	1333	1184	1343	1252
Western fir-spruce	113	5	13	33	2	9	9
Douglas fir	401	371	429	441	209	443	448
Coastal spruce-hemlock-redwood	37	104	223	58	41	98	92
Western pines	307	308	426	305	215	367	373
Western hardwoods	32	119	250	34	147	165	156
Moist tropical forest	0	130	77	118	19	55	24
Dry tropical forest	75	153	244	217	453	155	102
Oak hickory woodland	338	308	634	352	148	148	124
SE mixed woodland	188	113	366	214	770	72	91
Chaparral	39	80	40	30	143	42	73
Pinyon juniper	250	381	286	326	231	356	370
Total forest area	4126	4391	4446	4080	3645	4799	4805
Non-forest area	3579	3314	3259	3627	4061	2910	2904

water vapor leading to increases in water use efficiency of 30 to 40 percent. However, some species have been documented to acclimate to elevated CO_2 concentration by downregulating their photosynthesis (Bazzaz 1990; Gunderson and Wullschleger 1994; Wullschleger et al. 1997; Teskey 1997). On the other hand, most of the early research on effects of CO_2 was done on juvenile trees in pots and growth chambers. Evidence now shows that acclimation may not be as prevalent when roots are unconstrained (Eamus 1996; Curtis 1996). Moreover, limiting supplies of nutrients and water tend to only slightly restrict the growth response of trees to elevated CO_2 concentrations (Wullschleger et al. 1997).

While results from controlled exposure studies on seedlings and young trees are useful in describing the response of individual trees, they can only provide guidance on how such data can be extrapolated to the scale of mature trees, forest stands, and ecosystems. Simulating natural forest response to elevated CO_2 concentrations remains a challenge to the scientific community (Wullschleger et al. 1997). There are no data from which to assess the effect of elevated CO_2 on stand-level questions of LAI, and few data sets on tree responses can support a detailed analysis of growth per unit leaf area (Wullschleger et al. 1997). Forests could produce more leaf area under elevated CO_2 concentration, but this would increase transpiration and stand water use. Elevated temperatures would increase transpiration even further, possibly inducing a drought effect on the system by drying up the soil (Eamus 1996). This negative feedback would then reduce leaf area. These complex interactions are difficult to implement in the models, and each biogeography model includes its own simplified view of how the system might behave (for example, Neilson and Drapek 1998).

McGuire and Joyce (1995) summarized CO_2 effects on trees and therefore incorporated increased gross primary production in a biogeochemistry model used to evaluate the implication of climate change on U.S. temperate forests. In MAPSS, a decrease in stomatal conductance is assumed under elevated CO_2, which leads to enhanced water use efficiency and results in an LAI adjustment. Several studies have documented this enhancement for many tree species (Norby et al. 1986; Norby and O'Neill 1989; Oberhauer et al. 1985; Rogers et al. 1983a and b; Teskey and Shrestha 1985; Hollinger 1987). This effect is particularly important in regions where trees are more limited by moisture than nutrient availability (Conroy et al. 1986; Gifford 1979; Hollinger 1987; Idso 1989; Kimball and Idso 1983; McGuire et al. 1993; Polley et al. 1993; Sionit et al. 1980; Tolley and Strain 1984, 1985). It explains why temperate mixed forests in the eastern United States, for example, subject to a warmer and drier environment, are projected to decrease by 40 to 82 percent when stomatal conductance is left unchanged. They are only supposed to decrease by 12 to 14 percent or even increase by up to 9 percent, in the case of OSU scenario, when stomatal conductance is decreased by 35 percent (table 2.6). Because of the different ways models implement CO_2 effects, they can produce widely different simulations for the same climate change scenarios. Because MAPSS is very sensitive to changes in stomatal conductance and LAI, it predicts the most dramatic changes in moisture availability for all scenarios. Similarly, since BIOME2 includes an effect of CO_2 on the competitive balance between C3 and C4 plants, it simulates an expansion of C3 over C4 grasslands under milder climate change scenarios predicting only small increases in temperature (not shown here, VEMAP Members 1995).

Water Budget

Water use by vegetation is a complex interaction between lifeform water use efficiency, soil characteristics, snow dynamics, and climate (Dale 1997). It is thus difficult to predict a general response of how it will be affected by climate change. With increased temperatures and longer growing seasons, vegetation will transpire more water, thus making less water available in the form of runoff for irrigation or domestic uses. Thus, it is not surprising that in large areas of the conterminous United States, MAPSS simulates a decrease in runoff under all climate change scenarios, and in some regions quite drastically (Neilson and Marks 1994). For example, at the 10 km resolution in the tundra and taiga-tundra area, MAPSS simulates large decreases in runoff (60–86 percent of the total area undergoes a decrease in runoff) under both the GFDL-R30 and the Hadley Centre HADCM2SUL scenarios (tables A3 and A4 in Appendix A) that correspond to large increases in LAI (tables A5 and A6). Similarly, in grasslands, the decrease in runoff (61 percent of the total area) is due to an increase in LAI (80 percent of the total area) (tables A3, A4, A5, A6). MAPSS simulates significant areas (30–60 percent of the total area) of decreased runoff for temperate mixed forests (eastern United States), which are very sensitive to water losses. MAPSS also simulates large areas (80–100 percent of the total area) of increased runoff for temperate evergreen forests.

Simulation Uncertainties

Current Climate Source

The Cramer and Leemans dataset, derived from Leemans and Cramer (1991), used fewer U.S. stations and a different precipitation interpolation than the VEMAP dataset. As a result, the North American vegetation simulations do not capture mountainous vegetation as well as simulations using the VEMAP or VEMAP-derived 10 km datasets. Moreover, some distortion of vegetation boundaries also result from the different climatology.

Climate Scenario

Considerable uncertainty remains in the differences among the GCM climate scenarios (IPCC 1996; Ciret and Henderson-Sellers 1997). However, the capabilities of GCMs have improved significantly from the older (IPCC 1990) to the newer (IPCC 1996) scenarios, resulting in lower estimates of climate sensitivity. Nevertheless, some of the improvements—such as the inclusion of a cooling effect by aerosols—may prove to be less important than assumed by the climate modelers (Taylor and Penner 1994). Scientists generally agree on the likely rise in the average global temperatures over the next century, and that annual worldwide precipitation and evaporation will increase a few percent for every degree of warming. However, projections of climate change in specific areas are not forecasts but are reasonable examples of how the climate might change. By analyzing different scenarios from several different GCMs, the objective is to include a wide range of scientific uncertainty. But it is important to remember that climatic inputs derived from the GCMs have been interpolated to the higher resolution and may not correspond to realistic regional simulations.

Atmospheric circulation is strongly affected by fluxes of energy and water from the land surface. These fluxes depend on vegetation characteristics such as albedo, LAI, and vegetation height. Changes to land surface characteristics eventually feed back to the atmosphere. The current generation of climate models include the biophysical interactions between land surface and atmosphere in a “land surface module.” Sensitivity studies have now shown the importance of the feedback processes (for example, Bonan et al. 1992; Xue and Shukla 1993; Betts et al. 1997; Foley et al. 1998) and that using a fixed geographic distribution of vegetation types limits their use in global change studies. Unfortunately, all the assessments to date have been using ecosystem models that simulate changes in vegetation structure, with no feedback to climate models that produce the climate change scenarios they are so dependent upon. Results to date must thus be taken with caution since the atmosphere is totally decoupled from the land surface changes. Current research is now focusing on coupling fully dynamic representations of terrestrial ecosystems with climate models. Some of the new generation of biogeography models - dynamic global vegetation models or DGVMs - have already been designed to be fully coupled with climate models (for example, Foley et al. 1998).

Table 2.9—Predicted percent of the total area of North America and of the conterminous United States occupied by the various simplified vegetation types and percentage change when the GFDL-R30 scenario is applied and the CO_2 effect is included in MAPSS. Results correspond to three projects: 1) the North American project, from the Mexican border to northern Canada, at a half degree latitude-longitude resolution; 2) the VEMAP project at the same resolution as the North American project but concentrating on the continental U.S.; 3) the last project concentrating on the conterminous U.S. at a 10 km resolution. Percentage change in vegetation type area is calculated as: (scenario – current)/current. (*: in VEMAP the category taiga-tundra was not used since the BIOME model did not separate it from the boreal forest component.)

	Current climate			GFDL-R30		
Model resolution	half degree - North America	half degree - VEMAP (U.S.)	10 km - U.S.	half degree - North America	half degree - VEMAP (U.S.)	10 km - U.S.
Vegetation classes						
Tundra	16	<1	<1	–58	–100	–96
Taiga – Tundra	16	*	1	–39	*	–87
Boreal coniferous forest	16	2	1	+50	–100	–90
Temperate evergreen forest	7	9	6	+30	+142	+49
Temperate mixed forest	16	34	30	+7	–12	–36
Savanna woodland	12	12	16	+44	+15	+77
Shrub woodland	7	12	14	–20	–25	–20
Grassland	9	27	28	+23	–17	+5
Arid land	2	4	5	–64	–57	–47

Spatial Resolution

We compared simulation results from MAPSS at two different scales (10 km and half degree latitude-longitude resolution) and for two regions (North America and the continental United States). First, we compared results from 10 km resolution runs and results from VEMAP at approximately 50 km resolution (table 2.9 and fig. 2.3) for the continental United States using the GFDL-R30 scenario. The CO_2 effect was included in MAPSS. A small area increase (5 percent) in grasslands is simulated at the 10 km scale while a decrease of 17 percent is simulated at the VEMAP scale. Larger changes in the extent of temperate evergreen forests are simulated at the VEMAP scale with smaller changes in the extent of savannas. Simulation results agree in the direction of change for runoff pattern in the United States, with the exception of tundra areas, and anticipate increases in runoff in forest areas, savannas, and shrublands (table A3). Decreases in runoff are simulated in grasslands and arid lands. The model also simulates an increase in LAI (index of vegetation density) in tundra areas, savannas, shrublands, and arid lands, but a decrease in LAI in temperate mixed forests (table A5).

Secondly, we compared results obtained for the North American region (including Canada and the United States) and for the continental U.S. (VEMAP), both at a half degree latitude-longitude resolution. Changes due to the climate change scenarios are generally consistent both for runoff and LAI estimates (tables 2.9, A3, and A5).

Finally we compared results obtained for the North American region at a half-degree latitude-longitude resolution with those obtained for the continental United States at a 10 km resolution. There is good agreement between simulations except for the tundra area since coarse resolution results reflect large changes occurring in Canada but not in the United States (table 2.10). Tables A4 and A6 illustrate the agreement between simulations of LAI and runoff changes. The only disagreement occurs in the tundra and the boreal forest areas.

In summary, results from climate change impact simulations have to be carefully analyzed, the area of interest must be well delineated, and the scale of resolution specified. Trends in the expansion or reduction of certain systems can change dramatically between regions. Small changes that can be captured at high resolution can collectively modify the direction of trends. Caution is needed when analyzing model results for coarser resolution regional simulations.

Temporal Resolution Uncertainty

Current climate conditions used to run the models correspond to long-term average climate data that ignore extreme events and year-to-year variability. In reality, this variability greatly affects vegetation dynamics. Similarly, GCM-generated future climate scenarios for an "average" year are only snapshots of future climate at equilibrium with a doubled atmospheric CO_2 content. They do not

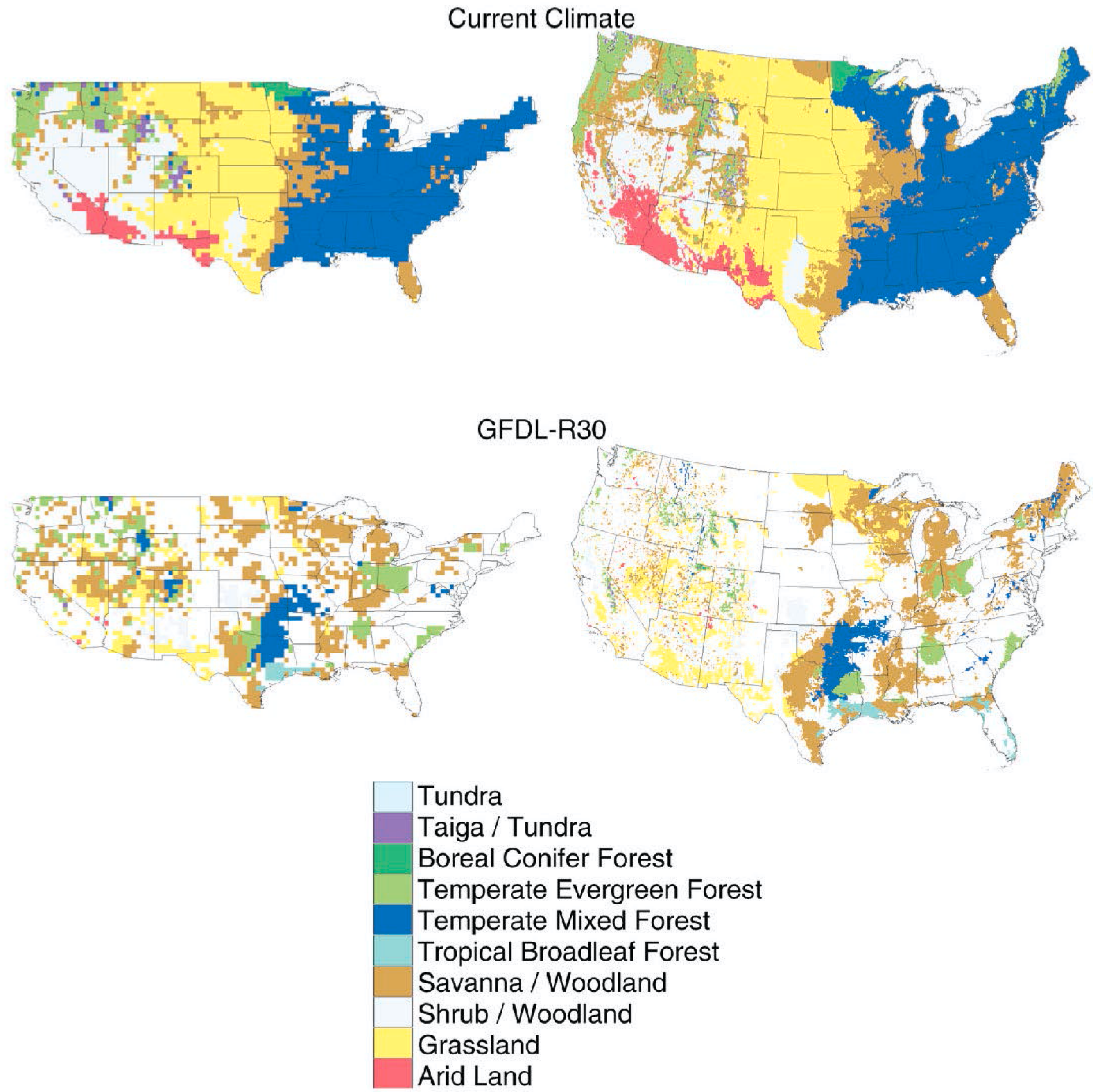

Figure 2.3—Aggregated potential vegetation classes simulated by MAPSS for current conditions (top) and areas where new vegetation classes are simulated by MAPSS using the GFDL-R30 climate change scenario (bottom) at a 10 km resolution (right) and at a half-degree resolution (VEMAP) (left) in the conterminous United States. Areas where there is no change in vegetation type remain white.

accurately represent the constantly evolving interactions between atmosphere, ocean, and land. In reality, there is no "average" year and thus equilibrium models such as MAPSS simulate vegetation distributions that do not and will not have an exact analog in nature (Borchers and Neilson 1998). The value of equilibrium projections, however, is that they depict theoretical equilibrium states or potential natural "climax" that the vegetation might evolve toward, a concept that has guided decision-making in forest management and silviculture for many years.

Table 2.10—Predicted percent of total area of either the North American region or the conterminous United States occupied by the various simplified vegetation types and percentage change in area when the Hadley Centre sulfate aerosol scenario (HADCM2SUL) is applied and the CO_2 effect is included in MAPSS. Results correspond to two projects using the MAPSS equilibrium biogeography model: one concentrating on the North American region from the Mexican border to northern Canada at a half degree latitude-longitude resolution, the other concentrating on the conterminous U.S. at a 10 km resolution. (+++ denotes a vegetation type that did not exist in the current climate scenario. NA corresponds to a vegetation type that did not exist in either current or future climates.) Percentage change in vegetation type area is calculated as: (scenario – current)/current.

	Current climate		HADCM2SUL	
Model resolution	half degree – North America	10 km - U.S.	half degree – North America	10 km - U.S.
Vegetation classes				
Tundra	16	<1	–37	–92
Taiga – Tundra	16	1	–41	–83
Boreal coniferous forest	16	1	+15	–99
Temperate evergreen forest	7	6	+82	+23
Temperate mixed forest	16	30	+52	+28
Tropical broadleaf forest	0	0	+++	NA
Savanna woodland	12	16	–1	–2
Shrub woodland	7	14	+1	+2
Grassland	9	28	–30	–18
Arid land	2	5	–70	–66

A new generation of models—the dynamic global vegetation models, or DGVM—is now emerging. These models couple vegetation structure and biogeochemical fluxes and simulate their dynamic changes as a response to changes in climate and disturbance regimes (Neilson and Running 1996; Foley et al. 1996; Friend et al. 1997; Lenihan et al. 1998). However, other constraints to the transient response of vegetation are still missing, such as soil development and seed dispersal. These models are being developed and should soon become the essential tools of future assessments.

Model Limitations

Nitrogen Budget

Nitrogen limitation is thought to moderate long-term responses to elevated CO_2 (Kirschbaum et al. 1994; McGuire et al. 1995; Eamus 1996). Climate change affects temperature- and moisture-controlled processes such as nutrient uptake, mineralization, and volatilization. Unless CO_2 stimulates an increase in nitrogen mineralization (Curtis et al. 1995; VEMAP Members 1995), productivity gains with high CO_2 concentration will be constrained by the available nitrogen (Körner 1995). Nitrogen limitations may constrain carbon gains to structural tissue rather than leaves (Curtis et al. 1995). Lifeform changes due to shifts in climate will also affect nutrient inputs (Pastor and Post 1988). Nitrogen fixation has been poorly quantified and has yet to be simulated accurately. Anthropogenic nitrogen fixation far exceeds natural nitrogen fixation (Vitousek 1994). In areas receiving large amounts of nitrogen deposition, a direct CO_2 response could result in large increases in leaf area. This increased LAI could increase transpiration and possibly provoke rapid soil water depletion, thus increasing the system sensitivity to drought. Nitrogen deposition has likely caused considerable accumulation of carbon in the biosphere since the last century (Vitousek 1994; Townsend et al. 1996). However, nitrogen saturation in soils can also be deleterious and possibly cause forest dieback in some systems (Foster et al. 1997). This effect is not included in MAPSS.

Disturbance

Disturbance intensity, frequency, and duration are likely to change with climate (Overpeck et al. 1990; Dale 1997). Natural fire frequency, duration, and intensity are closely tied to storm occurrences and precipitation regimes, which will be affected by global climate change (Dale 1997). Future climate coincident with changes in fire management practices and possible forest decline or dieback could bring longer fire seasons and potentially more frequent and larger fires in all forest zones (even those that do not currently support fire) (Fosberg 1990; Flannigan and Van Wagner 1991; King and Neilson 1992; Wotton and Flannigan 1993; Price and Rind 1994; Fosberg at al. 1996). Fire suppression during much of the 20th century has allowed biomass in many interior forests to increase considerably over historic levels (Agee 1990). With increased biomass, forests transpire almost all available soil water and become very sensitive to even small variations in drought stress. Forests are then highly susceptible to catastrophic fires even without global warming (Neilson et al. 1992; Stocks 1993; Stocks et al. 1996). Forests in the interior of North America are experiencing increased frequencies of drought stress, pest infestations, and catastrophic stand-replacing fires (Agee 1990). This sequence of events is a reasonable analog for what could happen to forests over much larger areas in the zones indicated by the biogeography models to undergo a loss of biomass or leaf area due to temperature-induced transpiration increases and drought stress (Overpeck et al. 1990; King and Neilson 1992). Because fire mediates rapid change, vegetation change could be significantly affected by changes in fire frequency.

The ability to predict changes in the frequency or intensity of extreme weather events such as drought, flooding, hail, hurricanes, and tornadoes using global and regional models is limited by their lack of small scale spatial and temporal resolution and uncertainties about representation of processes (IPCC 1996).

Conclusions

Current published assessments of biospheric responses to climate change are based on equilibrium models of the terrestrial biosphere such as MAPSS, BIOME2 or 3, and DOLY. These models simulate the combination of plant lifeforms that are in steady state with a given climate given a particular soil environment. These models, when run with various scenarios of climate change, show a series of strong responses:

1) Boreal forest and taiga-tundra regions are predicted to move northward or upward in elevation at the expense of the Canadian or alpine tundra (boreal forests are also simulated to experience diebacks of various degrees along the southern or lower elevational limits under all climate change scenarios). The boreal forest simulated in Minnesota for current climatic conditions totally disappears with even mild climate change scenarios such as the HADCM2SUL. Upper elevational or northern boundaries are predicted to shift upslope or northward.
2) Warmer scenarios produce the largest impacts on the boreal forest, but are also responsible for forest dieback in the conterminous United States.
3) Northwest and southeast forests might initially expand, then later contract in area. For the warmer climate change scenarios, MAPSS simulated extensive fragmentation of the eastern temperate mixed forests, which are very sensitive to changes in available water and thus to any positive effect of elevated CO_2.
4) Southwestern desert species may move into the Great Basin region given adequate thermal and hydrologic conditions.

The treatment of CO_2 effects in each of the three biogeography models strongly influences their simulations and explains some of the differences among them. Results from these equilibrium models also clearly depend on the climate change scenario that was used for the assessment. Although there is a growing consensus about the increase in future global average temperature, there is little agreement on the magnitude and timing of the changes in the hydrological cycle in various regions of the world. Moreover, large uncertainties remain about the regional changes of the various climate variables (Kattenberg et al. 1996). Therefore, assessments of future vegetation distribution carry the uncertainty intrinsic to the climate change scenarios and should not be considered as solid predictions. Equilibrium models, by definition, do not simulate dynamic or transient changes in vegetation assemblages. The rate of change predicted by the climate models may exceed historical rates of change (Kirschbaum et al. 1996). Simulation results may thus be used to indicate the direction of possible change, but not to estimate the time it might take a particular plant type to reach a new site (Cramer and Steffen 1997).

Movement of the various ecosystems may also be constrained after their initially rapid expansion by various factors such as lack of seed dispersal or establishment, lack of seasonal thermal requirements for establishment, poor soils, or unfavorable land use such as urbanization or cultivation. For example, the extent of the temperate mixed forest zone near the Great Lakes increases or declines depending, in part, on soil properties (Post and Pastor 1996). Moreover, vegetation types will probably not be displaced homogeneously. Different assemblages may appear and disappear over long periods of time (Huntley et al. 1997; Lenihan and Neilson 1995) and their composition will be strongly affected by changes in disturbance regimes.

Changes in boundaries limited by water balance are difficult to predict because of the complex interactions between changes in temperature, precipitation, and CO_2 concentration. Increases in rainfall are in some cases sufficient to balance increases in evaporative demand, and in other cases they are not. CO_2-induced changes in water use efficiency could reverse a potential drought response for certain plants. Northern states such as Minnesota, Michigan, and Wisconsin would endure displacement of forests by grasslands under all scenarios (figs. 2.2 and 2.3), a transition that would be mediated by drought and fire. Future climate changes coincident with changes in fire management practices and possible forest decline or dieback could bring longer fire seasons and potentially more frequent and larger fires in all forest zones. Drought and forest dieback could increase the fuel load and trigger more frequent and larger fires, while increased growth, given climatic oscillations, would also increase the fuel load. The importance of fire on vegetation change could increase and mediate rapid changes. Moreover, in the early stages of warming, when temperature increases are small, a CO_2-induced increase in water use efficiency could result in an expansion of temperate forests into neighboring drier areas and a concurrent increase in forest density throughout much of the current forest distribution. For example, MAPSS simulates the expansion of the temperate evergreen forest into Canada, where it replaces the taiga-tundra. It also simulates the expansion of the eastern temperate mixed forest westward into the central United States at the expense of savannas. A similar shift of northwestern forests into drier areas is simulated under the moderate warming scenarios. As the CO_2 effect saturates and temperatures continue to increase, however, the elevated evaporative demand could overwhelm the increased water use efficiency. Temperate forests could then contract in area and undergo a drought-induced decline in vegetation density (Neilson and Drapek 1998).

Comparing the "warmer" climate change scenarios with cooler ones illustrates what might happen to the southeastern mixed forest, where extensive fragmentation is simulated to occur with higher temperatures. The beneficial effects of elevated CO_2 could make a large difference in the response of the southeastern forests to the warming.

Finally, we want to emphasize that important factors, such as grazing by herbivores, invasions by weeds, diseases, and pests, and changes in land use due to human development, could drastically alter the responses of vegetation to climatic changes. There is currently no model that incorporates all these factors, and adding such complexity to currently existing models would also increase the margin of uncertainty in the resulting predictions. Climate change assessments should thus include these factors but new methods need to be developed to retain the usefulness of model simulations by keeping the uncertainty manageable.

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Appendix A

Table A1—Equivalence between VEMAP (VEMAP Members 1995) and MAPSS (Neilson 1995) vegetation categories and our simplified vegetation categories. E: evergreen, D: deciduous, B: broadleaf, N: needleleaf, PJ: pinyon juniper. C3 and C4 refer to the photosynthetic pathway of the plants.

VEMAP classes	Simplified categories	MAPSS categories
1. Tundra	1. Tundra	601. Tundra
2. Boreal coniferous forest	2. Taiga – Tundra	600. Taiga – Tundra
	3. Boreal coniferous forest	107. Forest EN taiga
3. Maritime temperate coniferous forest	4. Temperate evergreen forest	112. Forest EN maritime
4. Continental temperate coniferous forest	4. Temperate evergreen forest	108. Forest mixed warm EN
		113. Forest EN continental
5. Cool temperate mixed forest	5. Temperate mixed forest	102. Forest mixed cool
6. Warm temperate – subtropical mixed forest	5. Temperate mixed forest	101. Forest mixed warm DEB
7. Temperate deciduous forest	5. Temperate mixed forest	100. Forest deciduous broadleaf
		111. Forest hardwood cool
8. Tropical deciduous forest	7. Savanna woodland	
9. Tropical evergreen forest	6. Tropical broadleaf forest	105. Forest EB tropical
10. Temperate mixed xeromorphic woodland	7. Savanna woodland	210. Tree savanna PJ maritime
		109. Forest seasonal tropical ED
		110. Forest savanna dry tropical ED
		205. Tree savanna mixed cool EN
		206. Tree savanna mixed warm EN
		207. Tree savanna EN maritime
		208. Tree savanna EN continental
11. Temperate coniferous xeromorphic woodland	7. Savanna woodland	209. Tree savanna PJ continental
12. Tropical thorn woodland	10. Arid lands	305. Shrub savanna tropical EB
		309. Shrub savanna mixed warm EN
		404. Grass semi-desert
		425. Grass semi-desert C4
		500. Desert boreal
		501. Desert temperate
13. Temperate – subtropical savanna	7. Savanna woodland	200. Tree savanna DB
		201. Tree savanna mixed warm DEB
14. Warm temperate subtropical mixed savanna	8. Shrub woodland	310. Shrub savanna subtropical mixed
		303. Shrub savanna mixed warm DEB
		307. Shrub savanna mixed cool EN
		311. Shrubland subtropical xeromorphic
		313. Shrubland temperate conifer
		314. Shrubland temperate xeromorphic conifer
		423. Grass semi-desert C3
		424. Grass semi-desert C3-C4
15. Temperate coniferous savanna	7. Savanna woodland	211. Tree savanna PJ xeric continental
17. C3 Grasslands	9. Grasslands	414. Grass tall C3
		415. Grass mid C3
		416. Grass short C3
		418. Grass mid C3 C4
		419. Grass short C3 C4

continued

Table A1 (continued).

VEMAP classes	Simplified categories	MAPSS categories
18. C4 Grasslands	9. Grasslands	417. Grass tall C3 C4
		420. Grass tall C4
		421. Grass mid C4
		422. Grass short C4
19. Mediterranean shrubland	8. Shrub woodland	312. Shrubland subtropical mediterranean
20. Temperate arid shrubland	8. Shrub woodland	301. Open shrubland – no grass
		302. Shrub savanna DB
		308. Shrub savanna EN
21. Subtropical arid shrubland	10. Arid lands	502. Desert subtropical
		503. Desert tropical
		504. Desert extreme

Table A2—Equivalence between MAPSS assessment classes (Neilson and Chaney 1996), MAPSS (Neilson 1995) vegetation classes, VEMAP (VEMAP members 1995) classes, and our simplified vegetation classes. E: evergreen, D: deciduous, B: broadleaf, N: needleleaf, PJ: pinyon juniper.

Assessment classes	MAPSS classes	VEMAP classes	Simplified classes
Eastern forests			
1. NE mixed conifers and hardwoods	102. Forest mixed cool	5. Cool temperate mixed forest	5. Temperate mixed forest
2. NE mixed woodlands	205. Tree savanna mixed cool EN	10. Temperate mixed xeromorphic woodland	7. Savanna woodland
3. Maple-beech-birch	111. Forest hardwood cool	7. Temperate deciduous forest	5. Temperate mixed forest
4. Oak hickory forest	100. Forest deciduous broadleaf	7. Temperate deciduous forest	5. Temperate mixed forest
5. SE mixed pines and hardwoods	101. Forest mixed warm DEB	6. Warm temperate – subtropic mixed forest	5. Temperate mixed forest
Western forests			
6. Western fir - spruce	600. Taiga - Tundra	2. Boreal coniferous forest	2. Taiga – Tundra
	107. Forest EN taiga		3. Boreal coniferous forest
7. Douglas fir	112. Forest EN maritime	3. Maritime temperate coniferous forest	4. Temperate evergreen forest
	113. Forest EN continental	4. Continental temperate coniferous forest	
8. Coastal spruce - hemlock redwood	108. Forest mixed warm EN	4. Continental temperate coniferous forest	4. Temperate evergreen forest
9. Western pines	207. Tree savanna EN maritime	10. Temperate mixed xeromorphic woodland	7. Savanna woodland
	208. Tree savanna EN continental		
10. Western hardwoods	206. Tree savanna mixed warm EN	10. Temperate mixed xeromorphic woodland	7. Savanna woodland
Tropical			
11. Moist tropical forest	105. Forest EB tropical	9. Tropical evergreen forest	6. Tropical broadleaf forest
12. Dry tropical forest	109. Forest seasonal tropical ED	10. Temperate mixed xeromorphic woodland	7. Savanna woodland
	110. Forest savanna dry tropical ED		
Arid woodlands			
13. Oak hickory woodland	200. Tree savanna DB	13. Temperate - subtropical savanna	7. Savanna woodland
14. SE mixed woodland	201. Tree savanna mixed warm DEB	13. Temperate - subtropical savanna	7. Savanna woodland
Mediterranean			
15. Chaparral	312. Shrubland subtropical mediterranean	19. Mediterranean shrubland	8. Mediterranean fraction of shrub woodland
16. Pinyon juniper	209. Tree savanna PJ continental	11. Temperate xeromorphic coniferous forest	7. Savanna woodland
			7. Savanna woodland
	210. Tree savanna PJ maritime	10. Temperate mixed xeromorphic woodland	
			7. Savanna woodland
	211. Tree savanna PJ xeric continental	15. Temperate coniferous savanna	
Non forests			
17. Non forests	see Table A1	see Table A1	8. (Rest of) shrub woodland
			9. Grasslands
			10. Arid lands (see Table A1)

Table A3—Predicted percent area of increased or decreased runoff either in the North American region or the conterminous United States for the various simplified vegetation types when the GFDL-R30 scenario is applied and the CO_2 effect is included in MAPSS. Results correspond to three projects using the MAPSS equilibrium biogeography model: 1) the North American project, from the Mexican border to northern Canada, at a half degree latitude-longitude resolution; 2) the VEMAP project at the same resolution as the North American project but concentrating on the conterminous U.S.; 3) the last project concentrating on the conterminous U.S. at a 10 km resolution. Decrease corresponds to the areas where runoff decreases while Increase corresponds to areas where runoff increases.

	half degree – North America		half degree – VEMAP (U.S.)		10 km – U.S.	
Model resolution	Decrease	Increase	Decrease	Increase	Decrease	Increase
Vegetation classes						
Tundra	11	89	51	49	70	30
Taiga – Tundra	89	11	22	78	80	19
Boreal coniferous forest	0	100	0	100	0	100
Temperate evergreen forest	3	97	0	100	1	99
Temperate mixed forest	40	60	39	61	39	61
Savanna woodland	27	72	16	83	2	79
Shrub woodland	29	65	30	67	2	69
Grassland	61	35	69	30	69	30
Arid land	69	15	63	27	67	15

Table A4—Predicted percentage in area of increased or decreased runoff when the Hadley Centre sulfate aerosol scenario (HADCM2SUL) is applied and the CO_2 effect is included in MAPSS. Results correspond to two projects using the MAPSS equilibrium biogeography model: one concentrating on the North American region, from the Mexican border to northern Canada, at a half degree latitude-longitude resolution, the other concentrating on the conterminous U.S. at a 10 km resolution. Decrease corresponds to the areas where runoff decreases while Increase corresponds to areas where runoff increases.

	half degree – North America		10 km – U.S.	
Model resolution	Decrease	Increase	Decrease	Increase
Vegetation classes				
Tundra	18	80	60	40
Taiga - Tundra	76	24	86	14
Boreal coniferous forest	4	93	0	100
Temperate evergreen forest	17	81	1	99
Temperate mixed forest	44	56	32	67
Savanna woodland	37	58	37	60
Shrub woodland	17	81	13	85
Grassland	66	30	61	37
Arid land	43	42	36	51

Table A5—Predicted percent area of increased or decreased LAI either in the North American region or the conterminous United States for the various simplified vegetation types when the GFDL-R30 scenario is applied and the CO_2 effect is included in MAPSS. Results correspond to three projects using the MAPSS equilibrium biogeography model: 1) the North American project from the Mexican border to northern Canada at a half degree latitude-longitude resolution; 2) the VEMAP project at the same resolution as the North American project but concentrating on the conterminous U.S.; 3) the last project concentrating on the conterminous U.S. at a 10 km resolution. Decrease corresponds to the areas where LAI decreases while Increase corresponds to areas where LAI increases.

	half degree – North America		half degree – VEMAP (U.S.)		10 km – U.S.	
Model resolution	Decrease	Increase	Decrease	Increase	Decrease	Increase
Vegetation classes						
Tundra	0	62	13	87	11	88
Taiga – Tundra	1	97	0	100	0	100
Boreal coniferous forest	37	42	100	0	90	9
Temperate evergreen forest	20	47	31	67	49	41
Temperate mixed forest	88	10	66	32	85	13
Savanna woodland	19	73	14	81	15	80
Shrub woodland	17	76	6	90	11	83
Grassland	45	42	28	61	45	45
Arid land	2	92	0	94	0	91

Table A6—Predicted percent area of increased or decreased LAI when the Hadley Centre sulfate aerosol scenario (HADCM2SUL) is applied and the CO_2 effect is included in MAPSS. Results correspond to two projects using the MAPSS equilibrium biogeography model: one concentrating on the North American region, from the Mexican border to northern Canada, at a half degree latitude-longitude resolution, the other concentrating on the conterminous U.S. at a 10 km resolution. Decrease corresponds to the areas where LAI decreases while Increase corresponds to areas where LAI increases.

	half degree – North America		10 km – U.S.	
Model resolution	Decrease	Increase	Decrease	Increase
Vegetation classes				
Tundra	0	38	8	88
Taiga - Tundra	0	93	0	100
Boreal coniferous forest	14	54	61	35
Temperate evergreen forest	13	55	17	62
Temperate mixed forest	3	95	8	79
Savanna woodland	10	89	3	95
Shrub woodland	13	84	6	91
Grassland	3	93	14	80
Arid land	1	94	0	95

Ecosystem Productivity and the Impact of Climate Change

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Introduction

Earlier analyses of the supply and demand of timber assumed the continuation of historical climate and thus, did not explicitly incorporate factors such as temperature or precipitation into the projections of timber growth. Forests are adapted to local climates and changes in these climates are likely to impact future forest growth and timber outputs. Within the strategic planning process of the Forest Service (Joyce et al. 1997), the analysis of ecosystem productivity, as influenced by climatic factors, has been identified as a critical question in order to address the challenging problems associated with climate change:

- What are the likely effects of increasing atmospheric carbon dioxide and prospective climate change on ecosystem productivity, as measured by changes in net primary productivity?

In the last RPA Assessment (USDA Forest Service 1994), the consequences of elevated carbon dioxide and climate change on net primary productivity of forests were examined using climate model scenarios and an ecological model, TEM (Joyce et al. 1995). These results were then used to examine the impact of climate change on the supply and demand for timber products on private timberlands in the United States (see fig. 1.1, Joyce and Birdsey this volume). In this analysis, most of the forest productivity changes across the United States were positive, leading to increases in the timber inventories. With this change in harvestable inventories, timber harvests across the United States shifted as demand in various regions adjusted to take advantage of lower cost raw materials. Since these last RPA analyses, new experimental data and modeling analyses enhance the picture of the potential impact of carbon dioxide and climate change on forests. In this chapter, we present the results of such research.

New experiments on the impact of carbon dioxide on vegetation and meta-analyses of the accumulated research have demonstrated the impact of carbon dioxide on plant processes. In this chapter, we compare the impact of elevated carbon dioxide on ecosystem productivity, as determined from recent experimental data, with productivity results from the last RPA and other ecological model analyses. In addition, we explore whether these potential productivity shifts are within the range of productivity shifts that timber management treatments could induce in managed forests. We compare the results of economic research analyzing the potential to increase timber growth as an incentive to store carbon with these projected changes in productivity from climate change.

In the last RPA Assessment, two of the modeling assumptions were: 1) climate and vegetation were in equilibrium, and 2) the spatial scale of the ecological model was an adequate scale for national level analyses. Recent modeling studies have examined these assumptions.

In most ecological models (even now), vegetation is represented as pristine mature vegetation, rather than the actual vegetation of many different age classes, successional stages, introduced species, and management histories. Further, for both the ecological and the economic models, a broad range of habitats and species were aggregated into ecosystems or timber management types, respectively, for modeling purposes. The vegetation aggregation schemes and the nature of the spatial extrapolation differ between the ecological and the economic models. The implications of an ecological versus a timber management classification scheme are discussed in this chapter.

Computational problems arise when integrating or linking models that operate at different temporal and spatial scales. Computational limitations force a tradeoff between the spatial extent of the analysis (often dictated by policy considerations) and the grain of the analysis. Climate models, in order to compute global dynamics, operate at large spatial scales; grid cells range from 4 degrees to 10 degrees, resulting in coarse resolution of mountainous regions and small scale climate features. Meso-scale climate modeling now offers a finer depiction of climate features and the possibility to incorporate the effects of vegetation and land use feedbacks onto the atmospheric processes, but at large computational cost. Given this opportunity, it is important to understand the utility of going to a finer scale in the ecological analyses because the computational intensity increases by an order of magnitude when a 10 km scale is used instead of the traditional 0.5 degree scale. In this chapter, we report research results quantifying the climate change responses in ecosystem productivity at finer versus coarser spatial scales.

Plant and Ecosystem Productivity

Impact of Elevated Carbon Dioxide on Productivity

Net primary production (NPP) is the process by which the vegetation in an ecosystem captures carbon from the atmosphere. The changes in ecosystem productivity projected from ecological models reflect climate changes as well as the potential influence of carbon dioxide fertilization on net primary production. In the last RPA analysis (Joyce 1995), NPP of temperate forests in the U.S. increased from 8 percent to 27.2 percent, depending upon the climate scenario used (table 3.1). More recent analyses using a revised version of the Terrestrial Ecosystem Model (TEM) corroborate these earlier results, with some exceptions. A large modeling study compared the responses of several biogeochemistry models for all ecosystems within the conterminous United States to altered climate and elevated carbon dioxide (VEMAP members 1995). Averaged across all ecosystems, NPP responses increased from 1.7 to 34.6 percent (table 3.1). The NPP responses from TEM were higher than the responses for the Century Model (Parton et al. 1987, 1993) and the Biome-BGC model (Running 1994; Running and Coughlan 1988) (table 3.1). The largest NPP response for both TEM and Century (34.6 and 23.6 percent respectively) was for the UKMO scenario, in contrast to the response of Biome-BGC (1.7 percent increase). The lowest NPP increase for both TEM and Century was reported for the OSU scenario, again in contrast to the response of Biome-BGC. At the global scale, the NPP responses to climate change and elevated carbon dioxide for a revised version of TEM (Xiao et al. 1997) are lower than the NPP responses reported for the last RPA analysis. However, this result reflects the global extent of these measures.

These projected responses to altered climate and elevated atmospheric carbon dioxide incorporate the variation of climate and ecosystems across the United States or the globe. In contrast, experimental studies explore the underlying mechanisms for a plant's response to a change in atmospheric carbon dioxide in a controlled environment. Elevated atmospheric carbon dioxide has been shown to increase photosynthesis, enhance rates of carbon assimilation, increase stem and root biomass, and interact with other plant nutrients (Ceulemans and Mousseau 1994; Curtis and Wang 1998; McGuire et al. 1995b; Saxe et al. 1998). In a review of woody plants, Ceulemans and Mousseau (1994) reported the mean biomass increment from elevated CO_2 was +38 percent for coniferous trees and +63 percent for deciduous trees. For coniferous trees, the range of responses was from +0 percent to +95 percent. For deciduous trees, the range was from –47 to +290 percent. Summarizing studies not involving stress components, Saxe et al. (1998) reported larger average long-term biomass increment differences under elevated CO_2 for conifers of +130 percent and smaller averages for deciduous trees, +49 percent.

Table 3.1—Net primary production response (%) to climate change and elevated carbon dioxide for different biogeochemical models (TEM: Terrestrial Ecosystem Model; Cen: Century Model; BBGC: Biome-BGC) and at different spatial extents.

	Conterminous United States					Globe
	All forests		All ecosystems			Ecosystems
Climate scenario	TEM[1]	TEM[2]	TEM[3]	Cen[3]	BBGC[3]	TEM[4]
GISS[5]	27.2					20.6
GFDL-1	8.0					
GFDL-Q	12.1	13.1				18.5
OSU	17.4	29.6	26.5	14.6.	9.4	
GFDL-R30			30.5	22.1	20.2	
UKMO			34.6	23.6	1.7	
MIT L-O						17.8

[1] Joyce (1995)
[2] Nungesser et al. (1999)
[3] VEMAP members (1995)
[4] Xiao et al. (1997)
[5] GISS refers to the scenario from the Goddard Institute for Space Science model, GFDL-1 and GFDL-Q refer to results from the Geophysical Dynamics Lab model, OSU refers to a climate model developed by Schlesinger and others at Oregon State University, UKMO refers to the United Kingdom Meteorology Office model, and MIT refers to the Massachusetts Institute of Technology model

These responses from experimental studies are not without controversy. Of particular concern is whether the response is sustained over the life span of the tree. Most of these studies are done with seedlings or juvenile trees. Norby et al. (1992) concluded that a response in productivity was the result of an early stimulus and that no further sustained response was observed. Gorissen et al. (1995) suggest that an initial growth stimulation may be canceled by later physiological or morphological adaptations. For Yellow-popular (*Liriodendron tulipifera L.*), whole-plant carbon storage did not increase even though leaf-level photosynthesis and lower rates of foliar respiration in CO_2 enriched trees was observed (Norby et al. 1992). A number of studies have suggested that there may be a response specificity among tree genera (Ceulemans and Mousseau 1994) to an increase in atmospheric CO_2 as well as within genera (Ceulemans et al. 1996). DeLucia et al. (1994) suggested that allocation patterns in ponderosa pines may offset any increases in photosynthesis, resulting in potential declines in productivity under altered climate and elevated atmospheric carbon dioxide. Responses in natural stands are uncertain.

Curtis and Wang (1998) conducted a meta-analysis of over 500 reports on experiments on the effect of elevated carbon dioxide on woody plant mass, form, and physiology. These studies showed substantial variation in plant response to elevated CO_2, ranging from inhibition of growth to over 500 percent enhancement relative to plants grown in ambient conditions. Irrespective of the growing conditions, they found that total biomass increased significantly at about twice ambient atmospheric carbon dioxide concentrations, averaging a 31 percent increase. Stress altered the responses. Low nutrient availability reduced the CO_2 response to a 16 percent increase. Low light increased the response to 52 percent. They found no shifts in biomass allocation under elevated CO_2. Belowground responses were sensitive to length of the study and the stresses induced.

Pan et al. (1998) examined the modeled responses of terrestrial ecosystems to elevated atmospheric carbon dioxide. The forested ecosystem NPP response ranged from +3 to +23 percent increases (table 3.2). While these biogeochemistry models assume optimal responses similar to those observed experimentally (e.g., 25–50 percent), these spatially extrapolated responses to elevated carbon dioxide by ecosystem are substantially lower. When examined for underlying differences, Pan et al. (1998) noted that the three models tend to agree in their projected estimates of NPP response to doubled carbon dioxide along precipitation gradients, but differ along temperature gradients. Although the experimental literature is expanding with CO_2- impact studies, there is little information on the relative ecosystem-level response of NPP to elevated CO_2 along climatic gradients (Pan et al. 1998). These biogeochemistry models serve as different hypotheses on how ecosystem processes control the NPP response to elevated CO_2.

Table 3.2—Net primary production (NPP) response (%) to doubled atmospheric CO_2 (710 versus 355 ppmv) simulated by the VEMAP biogeochemistry models (Pan et al. 1998).

Forest type	BIOME-BGC	Century	TEM
Boreal conifer	6.05	3.37	3.50
Maritime conifer	10.09	4.10	7.59
Continental conifer	15.97	4.51	4.12
Cool temperate mixed	12.74	1.88	3.08
Warm temperate/subtropic mixed	6.69	2.25	11.82
Temperate deciduous	15.50	4.16	8.19
Temperate mixed xeromophic	10.94	10.00	21.22
Temperate conifer xeromophic	22.59	4.95	23.31

When experimental studies (since 1993) reporting changes in biomass are grouped by forest type, the species response is variable (table 3.3). This variability is explained, in some cases, by the different treatments. Optimal conditions, such as high N, tend to improve the biomass response to elevated carbon dioxide.

Within forest types, NPP responses (table 3.4) from the last RPA analysis ranged from a 0.9 percent decline for temperate deciduous forest productivity to an increase of 38.6 percent for boreal forest productivity. The experimental studies on woody species associated with boreal forest types showed responses to elevated carbon dioxide of 13 to 50 percent increases (table 3.3). Results from the modeling studies, which include potential changes in climate as well as carbon dioxide, ranged from increases of 23.8 to 38.6 percent (table 3.4). For the temperate deciduous species, the experimental results included a decline of 16 percent to an increase of 224 percent. Results in the modeled studies for temperate deciduous forests ranged from a decline of 0.9 percent to an increase of 36.6 percent. For conifer species, the experimental results ranged from no significant increases to an increase of 225 percent. Responses from the modeled studies for temperate coniferous forests ranged from a 15.7 percent increase to a 48.3 percent increase in NPP. These projected responses to elevated CO_2 and climate in the last RPA analyses are lower than the potential responses in the experimental studies (table 3.3 versus table 3.4).

Pan et al. (1998) detected the different ecosystem-level hypotheses that these biogeochemistry models reflect. These areas of uncertainty, if examined, identify opportunities to refine our ability to assess the impact of climate change on ecosystems:

- What role does the hydrological cycle play in controlling the CO_2 responses of leaf area and soil moisture along temperature and moisture gradients?

Table 3.3—Biomass response (percent) by woody species under elevated carbon dioxide in experimental studies.

Species	Significant	Non-significant	Percent Response	Source
Boreal				
Picea abies	Above-ground biomass (fresh wt)		16	Polle et al. (1993)
Picea glauca	Total biomass		44	Yakimchuk and Hoddinott (1994)
Picea mariana	Total biomass		13	Lord et al. (1993)
Picea mariana	Total biomass		50	Yakimchuk and Hoddinott (1994)
Picea sitchensis	Total biomass-irrigation/fertilization		52	Townend (1995)
Picea sitchensis		Total biomass-irrigation	19	Townend (1995)
Picea sitchensis	Total biomass-fertilization		44	Townend (1995)
Picea sitchensis	Total biomass-no irrigation or fertilization		49	Townend (1995)
Temperate coniferous				
Pinus banksiana		Total biomass	82	Yakimchuk and Hoddinott (1994)
Pinus silvestris		Shoot biomass	NS	Ineichen et al. (1995)
Pinus silvestris	Root biomass		57	Ineichen et al. (1995)
Pinus taeda		Total biomass-low N	37	Griffin et al. (1995)
Pinus taeda	Total biomass-high N		82	Griffin et al. (1995)
Pinus taeda	Total root biomass		124	King et al. (1996)
Pinus taeda	Total root biomass		225	King et al. (1996)
Pinus taeda	Total root biomass		64	King et al. (1996)
Pinus taeda	Total root biomass		102	King et al. (1996)
Pinus taeda	Total biomass		111	Tissue et al. (1996)
Pinus ponderosa		Total biomass-low temp	6	Delucia et al. (1994)
Pinus ponderosa	Total biomass-high temp		30	Delucia et al. (1994)
Pinus ponderosa		Total biomass-low N	48	Griffin et al. (1995)
Pinus ponderosa	Total biomass-high N		82	Griffin et al. (1995)
Pinus ponderosa	Total root biomass		97	King et al. (1996)
Pinus ponderosa	Total root biomass		86	King et al. (1996)
Pinus ponderosa	Total root biomass		153	King et al. (1996)
Pseudotsuga menziesii		Total biomass-age 3	37	Gorissen et al. (1995)
Pseudotsuga menziesii		Total biomass-age 4	3	Gorissen et al. (1995)
Temperate deciduous				
Prunus avium	Total biomass-low N		12	Wilkins et al. (1994)
Prunus avium	Total biomass-decline		-13	Wilkins et al. (1994)
Prunus avium	Total biomass-high N		81	Wilkins et al. (1994)
Prunus avium	Total biomass-high N		57	Wilkins et al. (1994)
Prunus avium X pseudocerasus	Leaf, shoot-2 month	Leaf, shoot, root-10 month	51	Atkinson et al. (1997)
Quercus robur	Leaf, shoot-10 month		224	Atkinson et al. (1997)
Quercus rubra	Total biomass, leaf mass		121	Lindroth et al. (1993)
Quercus rubra	Total biomass		47	Miao (1995)
Alnus rubra	Total biomass		72	Hibbs et al. (1995)
Alnus rubra	Total biomass		59	Hibbs et al. (1995)
Populus deltoides x nigra	Stem volume		58	Ceulemans et al. (1996)
Populus deltoides x nigra	Total branch biomass		108	Ceulemans et al. (1996)
Populus deltoides x nigra	Total biomass of leaves		49	Ceulemans et al. (1996)

continued

Table 3.3 (continued).

Species	Significant	Non-significant	Percent Response	Source
Populus deltoides x nigra	Total biomass		49	Curtis et al. (1995)
Populus deltoides x nigra	Total biomass		25	Curtis et al. (1995)
Populus tremuloides	Total biomass, leaf mass		48	Lindroth et al. (1993)
Populus trichocarpa x deltoides	Stem volume		43	Ceulemans et al. (1996)
Populus trichocarpa x deltoides	Total branch biomass		81	Ceulemans et al. (1996)
Populus trichocarpa x deltoides	Total biomass of leaves		36	Ceulemans et al. (1996)
Acer rubrum		Total, fine/coarse root mass	6	Berntson and Bazzaz (1996)
Acer saccharum		Total biomass	44	Lindroth et al. (1993)
Acer saccharum		Total biomass	7	Noble et al. (1992)
Acer saccharum		Total biomass	103	Noble et al. (1992)
Betula alleghaniensis	Stem mass, root mass, leaf mass		94	Rochefort and Bazzaz (1992)
Betula alleghaniensis family *G*	Total biomass		51	Wayne and Bazzaz (1997)
Betula alleghaniensis family *W*	Total biomass		30	Wayne and Bazzaz (1997)
Betula alleghaniensis family *Y*		Total biomass	–16	Wayne and Bazzaz (1997)
Betula lenta	Stem mass, root mass, leaf mass		119	Rochefort and Bazzaz (1992)
Betula papyrifera	Total biomass, fine/coarse root mass		43	Berntson and Bazzaz (1996)
Betula papyrifera	Stem mass, root mass, leaf mass		52	Rochefort and Bazzaz (1992)
Betula populifolia	Stem mass, root mass, leaf mass		144	Rochefort and Bazzaz (1992)
Liriodendron tulipifera	Tap root		12	Norby et al. (1992)
Liriodendron tulipifera		Branches, leaves, bole	37	Norby et al. (1992)

Table 3.4—Comparison of projected changes in forest productivity under climate change and elevated carbon dioxide.

	TEM[1]				TEM[2]	
Forest type	GFDL-1	GFDL-Q	GISS	OSU	OSU	GFDL-Q
Boreal	38.6	34.6	35.9	24.5	23.8	30.9
Boreal forest wetland	39.0	26.1	29.6	25.8	19.3	23.5
Temperate conifer	24.1	21.1	26.5	15.7	35.3	48.3
Temperate deciduous	–0.9	4.2	36.6	18.8	29.9	7.5
Temperate mixed	7.9	14.4	21.8	14.5	27.4	9.3
Temperate broad-leaved evergreen	23.0	20.7	24.8	17.2		
Temperate forest wetland	–0.1	3.6	25.4	34.8	42.2	2.3
All forests	8.1	12.2	27.2	17.4	29.6	13.1

[1] Joyce (1995)
[2] Nungesser et al. (1999)

- What role does the nitrogen cycle play in the CO_2 responses of leaf area and leaf nitrogen content along temperature and moisture gradients?
- What is the relative role of changes in nitrogen requirements, allocation, tissue C to N ratios, and rates of decomposition in determining CO_2 responses along temperature and moisture gradients?
- What are the relative contributions and importance of interactions between the hydrological and nutrient cycles in controlling NPP responses to elevated CO_2?

Importantly, Pan et al. (1998) conclude that future studies should measure the fluxes and the pools of carbon, nitrogen, and water. A clear picture of both fluxes and pools is important in improving our understanding of the interactions among processes that control CO_2 responses of ecosystems. Our understanding of these processes is the basis for the development of policies on carbon sequestration options in forests.

Climate Versus Management Influences in Timber Productivity

The productivity shifts in the last RPA climate change analysis were a response to increased atmospheric carbon dioxide and changes in temperature and precipitation. The time period was 50 years. Are those productivity shifts similar to the biological potential of current U.S. forests? Or are those productivity shifts similar to increases seen under economic opportunities fostered by timber management over a similar time frame? Vasievich and Alig (1996) used forest inventory data to assess the potential to increase timber growth for carbon storage. Biological opportunities were defined as the potential net annual growth of the most productive plots (top 20% of measured plots) for each site class, forest management type, and treatment opportunity on timberland suitable for treatment. This estimate represents actual management being applied to current stands, and was thus deemed achievable. Economic opportunities were defined as increases in growth on timberland that could be treated and yield 4% or more on the direct costs of treatment.

Based on Vasievich and Alig's (1996) analysis, biological opportunities exist to increase timber growth by about 8.6 billion cubic feet over 202 million acres, an increase of 39 percent over the current net annual growth of 22 billion cubic feet. Several decades would be required to implement the treatments to attain these increases. For economic opportunities, Vasievich and Alig (1996) estimated that net annual growth could be increased by 5.8 billion cubic feet, approximately 25 percent of the current net annual growth. Capital investment costs would be $10.9 billion. These biological and economic opportunities would take decades to implement, with the full effect not being seen until near the end of the 21st century. Thus, timber management could potentially enhance forest productivity to a larger degree (25 to 39 percent) than is currently projected for the productivity responses to changes in carbon dioxide or climate (8 to 29 percent, table 3.4).

Potential Vegetation and Current Vegetation Descriptions

In the last Forest Service climate change assessment, NPP response from the ecological model (TEM) was used to adjust growth in the forest sector model (TAMM-ATLAS-NAPAP) (fig. 1.1, Joyce and Birdsey this volume). The vegetation classification systems of these two models differed. Classification systems within the forestry sector have focused on commercial timber, while classification systems within botany and ecology have focused on the dynamics of pristine plant communities. Imbedded in the last RPA climate change assessment is the conversion of NPP responses from the ecosystem classification used in TEM into the forest management types used in the TAMM-ATLAS-NAPAP model.

Within the forest sector model, yield tables to project timber growth are derived from inventory plot data collected over a period of several years within each of the six forest inventory regions in the United States. One of the stratifications for these yield tables is timber types (table 3.5). The Forest Service inventory classifies forest land by forest types in which the named species, either singly or in combination, comprise a plurality of live tree stocking. The inventory types are based on a standard set of local forest types in the Forest Service Handbook, organized into broader forest type groups to facilitate reporting. There is some aggregation of the inventory forest types into the forest management types used in the TAMM-ATLAS-NAPAP model. The named species typically refers to a commercial tree species, for example Douglas fir, or to a class of fiber, such as softwood mix. The TAMM-ATLAS-NAPAP model was not developed to model geographically resolved data, hence the yield tables represented larger geographic regions, typically a multi-state grouping of ownership-forest type-age classes.

For the TEM model, similar to other biogeochemistry models, data from intensively studied ecosystems, representative of particular vegetation types, are used for calibration of model behavior. These models are spatially extrapolated by using vegetation maps. For the TEM model, the United States has been gridded into 0.5 degree by 0.5 degree grid cells. The Küchler classification system (Küchler 1964, 1978) has been used to assign the dominant potential natural vegetation (PNV) type to each

Table 3.5—Timber types used in the TAMM-Atlas forest sector model.

	Pacific Northwest			Rocky Mountain				
	West	East	Pacific Southwest	North	South	Northcentral	North	South
Douglas fir	X		X	X	X			
Douglas fir-mixed	X							
Douglas fir-larch		X						
Pure hemlock	X							
Fir-spruce	X	X		X	X			
True fir		X	X					
Pines	X							
Lodgepole pine		X			X			
Ponderosa pine		X	X	X	X			
Softwood mix	X							
Jack pine						X		
Red pine						X		
White pine						X		
White-red-jack pine							X	
Spruce-fir						X	X	
Red alder	X							
Redwood			X					
Hardwood mix	X							
Hardwood			X		X			
Mixed conifer			X					
Swamp conifer						X		
Oak-hickory						X	X	
Lowland hardwoods						X		
Maple-beech						X		
Loblolly-SRT-oak-gum							X	
Oak-pine							X	X
Elm-ash-red maple							X	
Maple-beech-birch							X	
Aspen-birch							X	
Planted pine								X
Natural pine								X
Upland hardwood								X
Lowland hardwood								X

grid cell (McGuire et al. 1992), and these types were then aggregated into a smaller set of ecosystem types, including 7 forest types: boreal forest, boreal forest wetland, temperate coniferous forest, temperate deciduous forest, temperate mixed forest, temperate broadleaf evergreen forest, and temperate forest wetland. This spatially explicit vegetation information is then used to extrapolate the ecological model to the larger spatial scale of the United States. In the last RPA analysis (Joyce 1995), NPP responses to climate change were computed for each 0.5 degree grid cell in the United States. Thus, NPP response data in TEM was resolved at a finer geographic scale than regional volume changes in the TAMM-ATLAS-NAPAP model.

A method was necessary to reclassify information from TEM so that this information could be used at the scale of the forest policy model. As the ecological and the timber classification systems had both been related to the PNV classification (McGuire et al. 1992, Garrison et al. 1977, Eyre 1980), this classification was used to link TEM ecosystem types with timber management types (table 3.6). The conversion process involved associating the TEM ecosystem type and the PNV type within each 0.5 grid cell (McGuire et al. 1992) to the forest type corresponding to that PNV type as defined by Garrison et al. (1977).

Several assumptions were made if no grid cell within a region was dominated by the PNV type associated with the forest management type. This situation might arise

Table 3.6—Translation of the Forest Classification used in the Forest Service FIA inventory using Potential Vegetation of Küchler (1964) to the VEMAP classification.

VEMAP vegetation	Kuchler[1]	FIA	TEM
Boreal coniferous (2)	15 Western spruce-fir (14)	Fir-spruce (17)	Boreal coniferous forest
	21 Southwestern spruce-fir (20)	Fir spruce (17)	
	93 Great Lakes spruce-fir (84)	Spruce-fir (2)	
	96 Northeastern spruce-fir (87)	Spruce-fir (2)	
	*97 Southeastern spruce-fir (88)	Spruce-fir (2)	
Temperate maritime coniferous forest (3)	1 Spruce-cedar-hemlock (1)	Hemlock-Sitka spruce (12)	Maritime temperate coniferous forest
	2 Cedar-hemlock-Douglas fir (2)	Douglas-fir (11)	
	3 Silver fir-Douglas fir (3)	Fir-spruce (17)	
	4 Fir-hemlock (4)	Fir spruce (17)	
	5 Mixed conifer (5)	Ponderosa pine (13)	
	6 Redwood (6)	Redwood (18)	
Temperate continental coniferous forest (4)	*7 Red fir (7)	Fir-spruce (17)	Continental temperate coniferous forest
	*9 Pine-cypress (9)	Ponderosa pine (13)	
	8 Lodgepole pine-subalpine (8)	Lodgepole pine (15)	
	10 Ponderosa shrub (none)	Ponderosa pine (13)	
	11 Western ponderosa (10)	Ponderosa pine (13)	
	12 Douglas fir (11)	Douglas fir (11)	
	13 Cedar hemlock pine (12)	Western white pine (14)	
	14 Grand fir Douglas fir (13)	Larch (16)	
	16 Eastern ponderosa pine (15)	Ponderosa pine (13)	
	17 Black Hills pine (16)	Ponderosa pine (13)	
	18 Pine Douglas fir (17)	Ponderosa pine (13)	
	19 Arizona pine (18)	Fir-spruce (17)	
	20 Spruce fir - Douglas fir (19)	Fir-spruce (17)	
	95 Great Lakes pine (86)	White-red-jack pine (1)	
Cool temperate mixed forest (5)	28 Mosaic of cedar hemlock-Douglas fir forest and Oregon oakwoods (24)	Douglas fir (11)	50% continental temperate coniferous forest, 50% temperate deciduous forest
	106 Northern hardwoods (97)	Maple-beech birch (9)	
	107 Northern hardwoods fir (98)	Maple-beech-birch (9)	
	108 Northern hardwoods spruce (99)	Maple-beech-birch (9)	
	109 Mosaic of Appalachian oak and Northern hardwoods		
	110 Northeastern oak-pine (100)	Loblolly-shortleaf (4)	
	106 Seral northern hardwoods (97)	Aspen-birch EAST	
	107 Seral northern hardwoods fir (98)	Aspen-birch EAST	
	108 Northern hardwoods spruce (99)	Aspen-birch EAST	

continued

Table 3.6 (continued).

VEMAP vegetation	Kuchler[1]	FIA	TEM
Warm temperate/subtropical mixed forest (6)	29 California mixed evergreen (25)	Douglas fir (11)	33% continental temperate coniferous forest, 33% temperate deciduous forest, 34% temperate broadleaved evergreen forest
	89 Blackbelt (80)	Oak-hickory (6)	
	90 Live oak sea oats (81)	Oak-gum-cypress (7)	
	*91 Cypress savanna (82)	Oak-gum-cypress (7)	
	*105 Mangrove (96)	Oak-gum-cypress (7)	
	111 Oak-hickory-pine (101)	Loblolly shortleaf pine (4) and oak-pine (5)	
	112 Southern mixed (102)	Longleaf-slash pine (3) and oak-pine (5)	
	*115 Sand pine scrub (105)	Loblolly-shortleaf (4)	
	*116 Subtropical pine forest (106)	Longleaf-slash (3)	
Temperate deciduous forest (7)	26 Oregon oakwoods (22)	Western hardwoods (21)	Temperate deciduous forest
	98 Northern floodplain (89)	Elm-ash-cottonwood (8)	
	99 Maple-basswood (90)	Maple-beech-birch (9)	
	100 Oak-hickory (91)	Oak-hickory (6)	
	101 Elm-ash (92)	Elm-ash-cottonwood (8)	
	102 Beech maple (93)	Maple-beech-birch (9)	
	103 Mixed mesophytic (94)	Maple-beech-birch (9)	
	104 Appalachian oak (95)	Oak-hickory (6)	

[1] Küchler numbering scheme. The first number in each box is the number reported in the 1964 Küchler publication. The number in () after the type name is the number reported on the 1978 map, which was also the number used in the 1997 Forest Service publication mapping FRES Ecosystems and the Küchler System.

In VEMAP, vegetation types represented by map symbols 7, 9, 22, 25, 27, 72, 73, 91, 97, 105, 116, 117 never dominated at the 0.5 longitude $\times$ 0.5 latitude grid cell representation. Types 27 Mesquite bosques, 72 Sea oats prairie, and 73 Northern cordgrass prairie are non-forest types. For these types, I used a table provided by Dave Kicklighter that had identified the TEM/VEMAP type associated with these Küchler types.

FIA types of Aspen-birch 10 and 22 are east and west expressions of the Aspen-birch type. In the East, this type was indexed to the seral stages of Northern hardwoods, Northern hardwoods-fir forests, and Northern hardwoods-spruce forest and identified as temperate deciduous VEMAP type. In the West, this type was also coded as temperate deciduous VEMAP type. All FIA western hardwoods type were identified as temperate deciduous VEMAP type.

if silvicultural management favored a seral species, or if climax vegetation were not dominant within parts of the region, or if the commercial species happened not to be the tree species named in the climax type. For example, softwood mix, a forest management type used in the Pacific Northwest-west timber supply region, was a mix of two forest types: redwood and larch. No PNV type was representative of redwood within the Pacific Northwest-west region (west side of Oregon and Washington). The nearest PNV type was the redwood type in the Pacific Southwest region. Here, the assumption was made that the response to climate change would be more appropriately described by using a similar ecosystem type, but outside of the region rather than a different vegetation but inside the region. In another example in the Pacific Northwest-west region, red alder was a forest management type that did not have a corresponding PNV type. According to Eyre (1980), red alder is a successional type replaced by the Pacific Douglas fir and western hemlock-sitka spruce types. Eyre (1980) did assign red alder to the inventory type of western hardwoods, and Garrison et al. (1977) assigned the PNV type Oregon oakwood with the inventory type of western hardwoods. However, this PNV type occurs solely along the Oregon-California border in the Pacific Northwest-west region, and red alder is common on bottom lands, sheltered coves, and on moist slopes of the Coast and Cascade ranges (Eyre 1980). Thus, the PNV type associated with the climax vegetation that typically replaces red alder, the cedar hemlock-Douglas fir type, was used to modify the yield table for red alder.

In the Pacific Northwest-east region, lodgepole pine was a forest management type used in the forest sector model. However the PNV type, lodgepole pine, did not dominate any of the grid cells within this region. According to Eyre (1980), lodgepole pine within this region was associated with subalpine fir, Engelmann spruce, white spruce, and Rocky Mountain Douglas fir. The PNV types with these associates (fir-hemlock, Douglas fir, western spruce-fir) were used to define the response of lodgepole pine within this region.

The aspen-birch type used as a forest management type in the Northeast region is considered by Küchler to be a seral vegetation, replaced by Northeastern spruce-fir. Eyre (1980) considered aspen-birch to be a boreal hardwood. Aspen will be replaced by the PNV types of northern hardwoods or spruce-fir types, and succession to these types is more rapid than to pine (Eyre 1980). The volume of aspen-birch was modified by the TEM NPP response from an aggregation of Northeastern spruce fir, northern hardwoods, and northern hardwoods-spruce PNV types.

In the Southern region, planted pine and natural pine were two forest management types (table 3.5). There were no corresponding PNV types for these forest types. For the last RPA analysis, the oak-hickory pine and the southern mixed forest PNV types were used to assess the impact of climate change on timber volume of the two forest management types.

Clearly there are limitations with assigning forest management types to potential natural vegetation types. Ecological models represent the most current understanding of how ecological processes operate at the ecosystem scale. The potential distribution of these ecosystems implies a similarity of ecosystem function within the range of each ecosystem type as well as a geographic presence unaltered by land use. Forest management models have focused on the yield of wood from forestland. The aggregation of inventory plots into a forest management type implies a similarity in timber production with the geographic range of that forest management type. For both of these classification systems, the ecosystem dynamics or timber production within a type may be quite variable. Current land uses have altered potential distributions. Inventory data is more likely to represent the current distribution of forests.

Use of vegetation types presents problems by potentially ignoring differences between species and new associations and how both may affect ecosystems under climate change. The assumption that species associations will remain constant is a consequence of lumping species-specific information into a "type" (Kirschbaum and Fischlin 1996). Species associations have been very different in the past under different climates. Davis (1989) reports paleoecological evidence of community types no longer present, such as the spruce-oak woodland association. The approach of using functional types (Henderson-Sellers and McGuffie 1995; Woodward et al. 1995) in lieu of species in climate change modeling is attractive because this approach reduces the computational complexity associated with projecting each plant species. However, the use of functional types often assumes that these groupings of species will remain together and respond to climate change as a unit. The use of functional types raises the issue of whether "functional types" preserve species differences (Solbrig 1994). The value of these functional types is that they group similar physiological and ecological roles (Solbrig 1994; Vinton and Burke 1995), but species behavioral differences may be overlooked. For example, is the rate of reaching equilibrium with climate the same for all species within a functional group?

Similarly, the aggregation of diverse tree species into forest management types presents problems by ignoring potential differences between commercial timber species, and how climate change might alter wood fiber production; for example, how volumes might shift, when wood develops, and the quality of wood under altered climate. Experimental results suggest that the responses to climate change might be genera, if not species, specific (see earlier discussion).

Within the last RPA analysis, the impact of climate change on NPP was assessed using the historical range of temperate forests (fig. 3.1), not the current distribution as affected by land use. Thus, the ecological variability analyzed under climate change represented a greater ecological amplitude than forests currently represent. The spatial distribution of existing forests using the forest management type classification has been recently mapped. We used this spatial distribution to develop a map of vegetation classified by the same system used by the VEMAP members. The forest management type information was available at the 1 km scale. We identified a Küchler PNV type for each grid cell on the forest management map, based on Garrison et al. (1977) (table 3.6). Once the Küchler type was associated with each grid cell, we then used the classification given in VEMAP members (1995) to link the cell to a VVEG type (table 3.6). The map was then resampled to the 10 km scale (fig. 3.2). The finer scale of this map allows smaller isolated patches of forest to remain on the map, particularly in the Southwest and in the Great Plains. However, this distribution contrasts with the distribution used in the last RPA analysis (fig. 3.1) in the drastically reduced area of forests in the eastern part of the United States, and in the patchiness of the forests across the United States. The area of forests in the Midwest region and the Mississippi River valley declines when the land use in agriculture is removed. The homogeneity of vegetation types is lessened in figure 3.2, particularly for the New England states, where boreal, temperate coniferous, and cool temperate mixed forest types intermingle in contrast to the uniformity in figure 3.1. This re-examination of forest types (fig. 3.2) could be used as the basis for an analysis of the impact of climate change on forest productivity. It would likely reflect the potential shifts in forest productivity more closely because climate shifts in regions of existing forests would be used as climate input to a model such as TEM.

Projecting Ecosystem Productivity at Different Spatial Scales

Climate, topography, vegetation, and soils input data for ecological models used in large-scale integrated assessments are typically gridded at the 0.5 degree longitude by 0.5 degree latitude scale (Cramer et al. 1999; Heimann et al. 1998; Kicklighter et al. 1999; Melillo et al. 1993; VEMAP members 1995). Use of gridded input data implicitly assumes that the mean or dominant surface features represent the entire grid cell. Inherent in this assumption is the uncertainty with which this gridded value represents the heterogeneity of the actual coarse grid cell features and the representativeness of this gridded value when used in the ecological models to describe the biological processes operating within the grid cell. While opportunities exist to move the analysis of the impact of climate change on forests to a finer grid scale, this reduction in grid size would increase the processing time by an order of magnitude. Hence it is important to assess the uncertainty that aggregation of climate data contributes to the estimation of forest productivity under climate change. Nungesser et al. (1999) used the Terrestrial Ecosystem Model (TEM version 4.0; McGuire et al. 1995a) to evaluate the utility of moving the climate change impact analysis from the 0.5 degree scale used in the last RPA analysis to the 10 km scale.

The effect of aggregation on the estimation of productivity has been studied. Net primary production (NPP) estimates differed by 20 percent when coarse grain versus fine grain soils data were used as input data for the PnEt model (Lathrop et al. 1995). Pierce and Running (1995) obtained overestimates of up to 30 percent in NPP from the FOREST-BGC model when sub-grid variations in climate, topography, soils, and vegetation were averaged across a series of grain sizes from 1 km to 1 degree. Most of this error was produced by average temperature, while average topography, soils, and vegetation types also contributed.

Nungesser et al. (1999) examined the impact of two different spatial resolutions on the simulated forest ecosystem responses for a baseline climate and two climate change scenarios. The TEM model uses spatially resolved information on climate (monthly precipitation, monthly mean air temperature, and cloudiness), soil texture (percent sand, silt, and clay), vegetation type, and elevation. The fine resolution grid cells were 100 km^2 in size, and 25 of these cells were nested within a coarse resolution grid cell of approximately a half degree in size (2500 km^2). The 10 km $\times$ 10 km raster data for climate (monthly precipitation, monthly mean air temperature) were obtained from Neilson (personal communication) as described in Daly et al. (1994), Marks (1990), and Neilson (1995). For the 50 km $\times$ 50 km input data, the fine scale input data sets for continuous variables were averaged to the 50 km $\times$ 50 km resolution. Averaging to the 50-km scale results in the smoothing out of precipitation and temperature values along the gradients of change and the loss of finer detail in some areas (fig. 3.3 and 3.4). Areas of fine scale patchiness of precipitation in the western mountains and the southern coastal plains are smoothed out at the 50-km grid scale (fig. 3.3). Annual average temperature values are influenced in areas where there is substantial temperature variability such as around the mountainous areas in the West (fig. 3.4)

The historical range of temperate forests was the spatial extent of the Nungesser study. Küchler vegetation

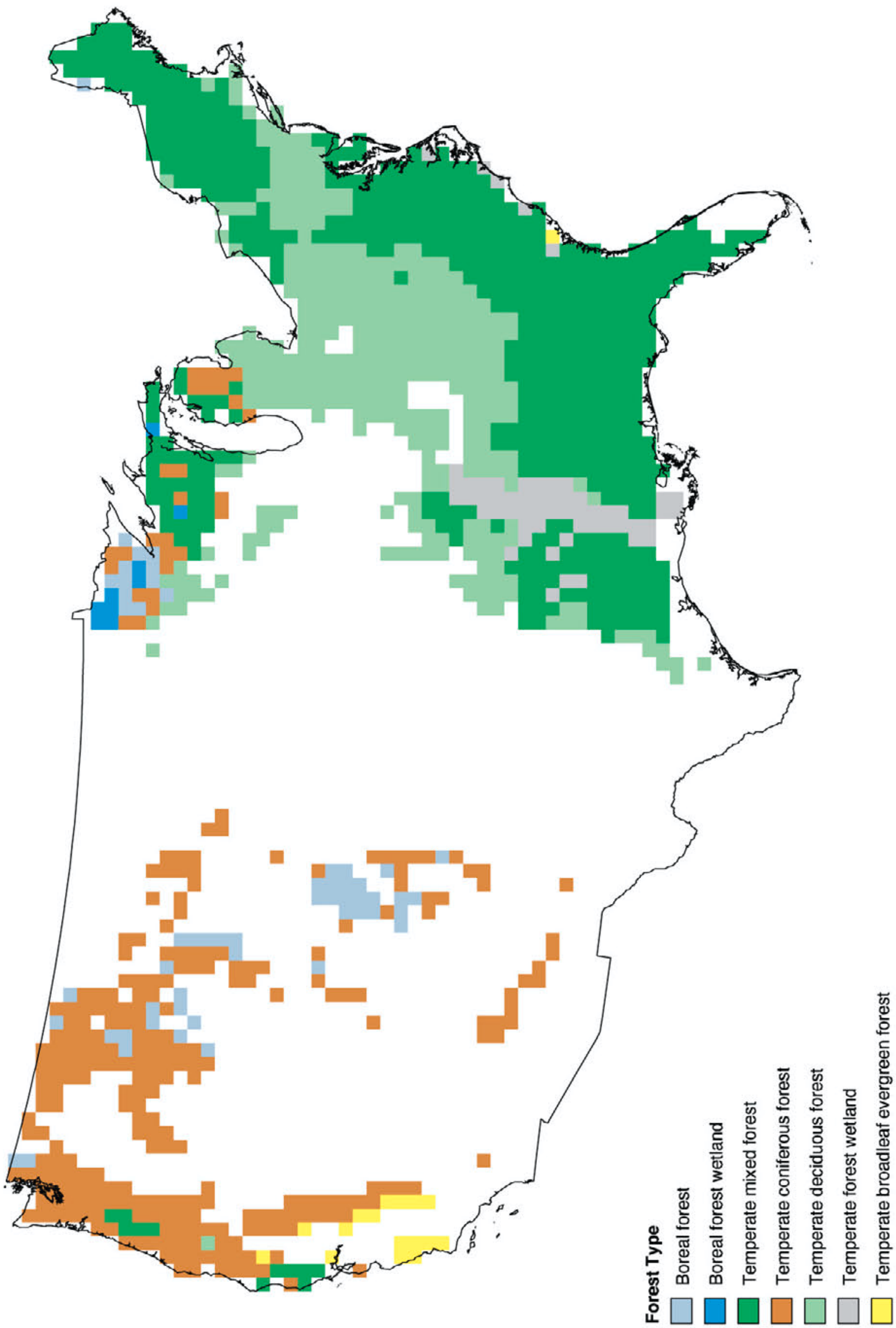

Figure 3.1—Forest type distribution used in the last RPA analysis (Joyce et al. 1995).

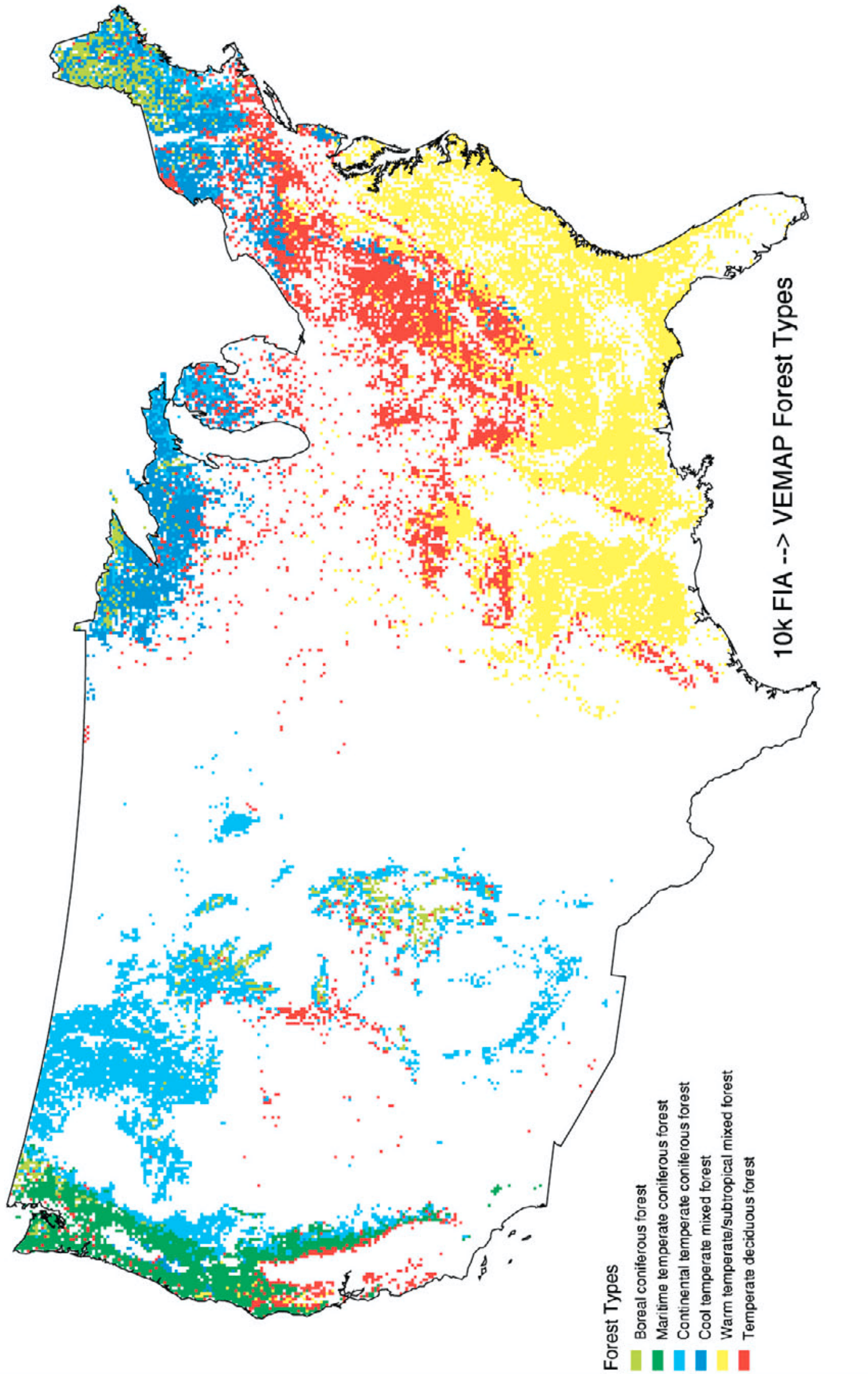

Figure 3.2—VEMAP vegetation classification associated with the current distribution of forests based on USDA Forest Service inventory plots.

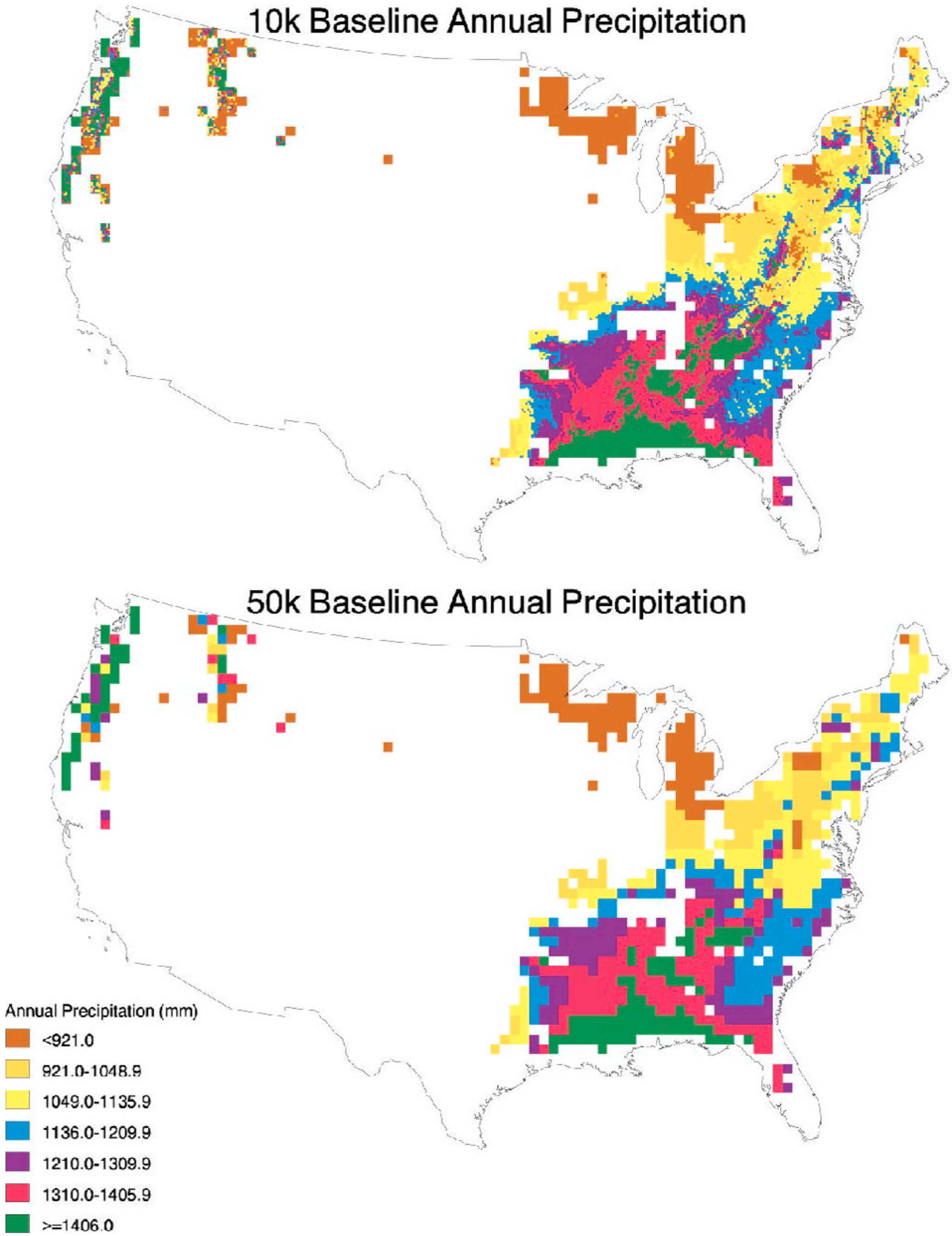

Figure 3.3—Baseline annual precipitation at 50 km scale and 10 km scales.

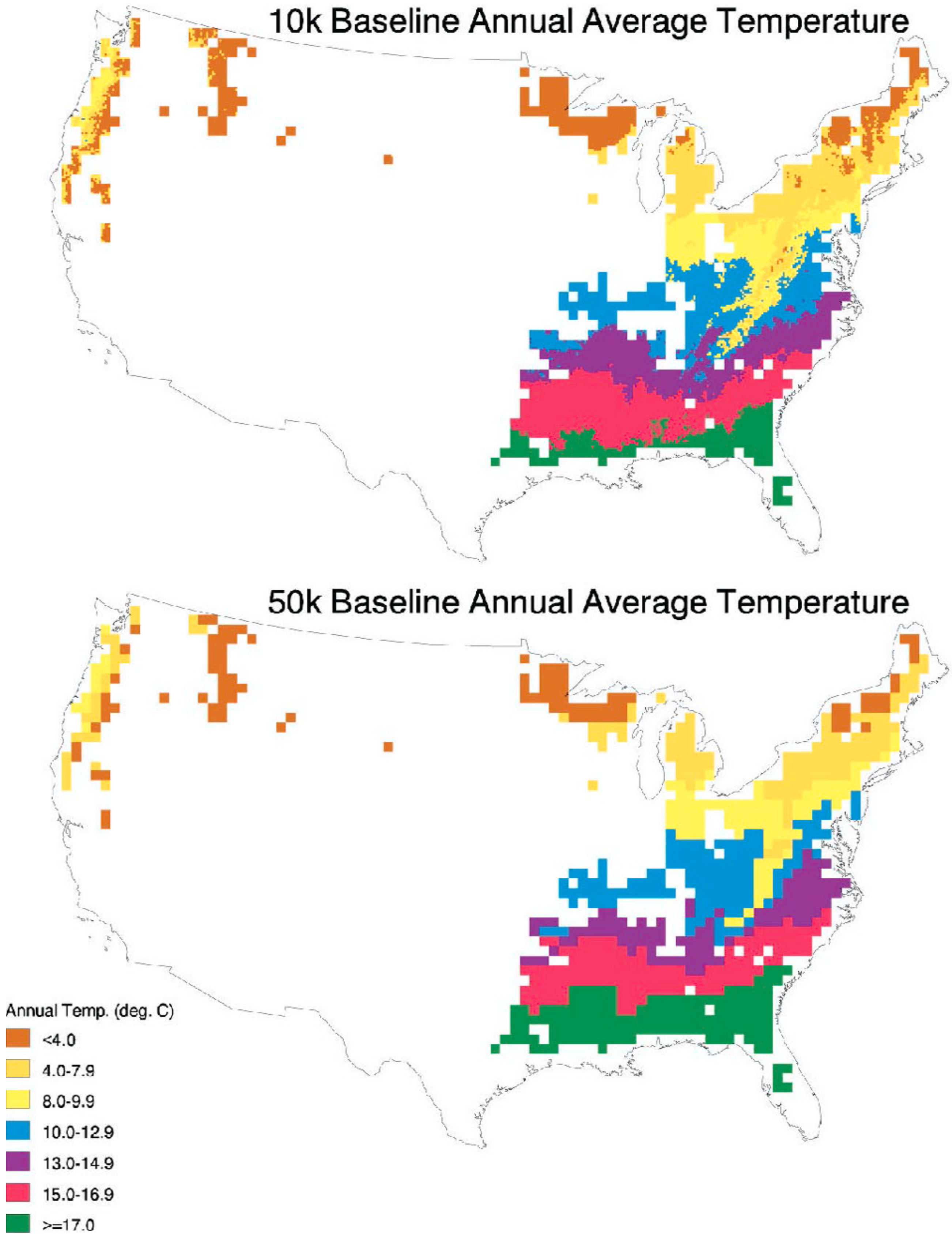

Figure 3.4—Baseline annual average temperature at 50 km and 10 km scales.

was digitized from the 1975 map (Küchler 1975) at 10 km resolution (Steve Hodge, personal communication) and was reclassified to TEM vegetation types. For the 50 km × 50 km input data, the fine scale input data set for vegetation were aggregated to the 50 km × 50 km resolution through the use of the majority rule (proportional aggregation method of Costanza and Maxwell 1994). While the coarse resolution grids include six vegetation types (fig. 3.5), a seventh type, temperate evergreen broadleaf, appeared in only three grid cells at the fine resolution. All forest types that comprised greater than 10 percent of the total area within the 10-km grids retained at least 93 percent of their total area when aggregated to the 50 km grid size. The rarer forest types, boreal, wet boreal, temperate forest wetland, and temperate broadleaf evergreen, lost from 28 to 100 percent of their area under the proportional aggregation rule. Rare types dispersed across the landscape were most likely to be lost in the aggregation. For example, the boreal grid cells in New England disappear at the 50-km scale (fig. 3.5). Similarly, some of the forested boreal wetland forests in the northern Midwest are lost at the 50-km scale.

An aggregation error was computed as the difference of the 10-km estimate of NPP and the 50-km estimate of NPP divided by the 10-km estimate of NPP. This aggregation error was computed for the baseline runs (baseline climate and CO_2 levels of 355 meq/l):

$$E_B = 100 * \left(\overline{NPP_{10}} - NPP_{50}\right) / \overline{NPP_{10}} \tag{1}$$

where E_B is the relative aggregation error and $d_{GQ50} =$ is the average of the mean annual NPP in gC/m^2/yr estimates for the 25 fine grid cells, and NPP_{50} is the estimate of the NPP at the coarse grid scale.

The NPP results for the 10 km reflect finer scale patterns than the 50 km results (fig. 3.6), but these patterns are not sufficient to generate large aggregation errors at the national, forest type, or grid cell scale. The NPP estimate for all forests at the national extent differed by less than 1 gC/m^2 between the fine and coarse resolution scales, 675.8 versus 676.7 gC/m^2 (table 3.7). Aggregation error based on these 815 grid cells is very small and negative (–0.4%). Estimates of NPP at the 50-km grid scale differed from the corresponding average for the 10-km grid cells by less than 10 percent across most of the historic range of temperate forests. The smallest aggregation error was found generally throughout the East and Southeast, as well as in the western mountains. Rarely were aggregation error differences greater than 20 percent at the individual grid cell level. These larger aggregation errors occurred around the Great Lakes, in northern New England, and in the Rocky Mountains. By forest type, the aggregation errors were still small, less than 2 percent. Estimates of NPP differed by less than 10 gC/m^2 in most cases (table 3.7). The largest aggregation error occurred in boreal and forested boreal wetland forests, with positive values of 11.8 and 9.6 percent, respectively. Estimates of NPP for boreal forest at the 50-km grid scale were less than the 10-km estimate by approximately 37 gC/m^2 (table 3.7).

Table 3.7—Net primary production (NPP) for baseline climate. Values are net primary production in gC/m2; reported values are means and standard deviations are in parentheses followed by ranges. The "(n)" is the number of coarse resolution grid cells in each forest type.

Forest Type (n)	Resolution	NPP	
Boreal (9)	Fine	312.4 (48.8)	206–380
	Coarse	275.2 (42.6)	163–297
Forested boreal wetland (4)	Fine	319.0 (36.6)	291–372
	Coarse	285.0 (9.8)	271–291
Mixed temperate (409)	Fine	696.1 (124.9)	339–893
	Coarse	691.0 (113.8)	346–863
Conifers (91)	Fine	344.7 (96.3)	158–540
	Coarse	349.4 (97.9)	181–551
Deciduous (260)	Fine	751.5 (66.3)	483–877
	Coarse	761.3 (60.7)	600–909
Temperate forested wetland (42)	Fine	838.5 (61.0)	751–1046
	Coarse	846.6 (62.0)	790–1058
ALL FORESTS (815)	Fine	675.8 (167.9)	158–1046
	Coarse	676.7 (165.3)	163–1058

The climate change scenarios (temperature, precipitation) at the 10 km scale were based on two GCMs used in the last RPA analysis: the Geophysical Fluid Dynamics Laboratory Q-flux (GFDL-Q) (Manabe and Wetherald 1987) and Oregon State University (OSU) models (Schlesinger and Zhao 1989). Using the same protocol as for baseline climate, fine resolution climate input data were averaged within a coarse grid to serve as coarse resolution inputs to TEM. Values for forest type, elevation, and soils remain unchanged from the baseline simulation. The climate change scenarios included a CO_2 concentration of 625 ppmv. Aggregation error in NPP for the two climate change scenarios was computed in the same manner as the baseline aggregation error.

Grid-level response of net primary productivity to the climate change scenarios were calculated as:

$$d_{OS10} = 100 * \left(\overline{NPP_{OS10}} - \overline{NPP_{10}}\right) / \overline{NPP_{10}}, \tag{2}$$

$$d_{OS50} = 100 * \left(NPP_{OS50} - NPP_{50}\right) / NPP_{50}, \tag{3}$$

$$d_{GQ10} = 100 * \left(\overline{NPP_{GQ10}} - \overline{NPP_{10}}\right) / \overline{NPP_{10}}, \text{ and} \tag{4}$$

$$d_{GQ50} = 100 * \left(NPP_{GQ50} - NPP_{50}\right) / NPP_{50} \tag{5}$$

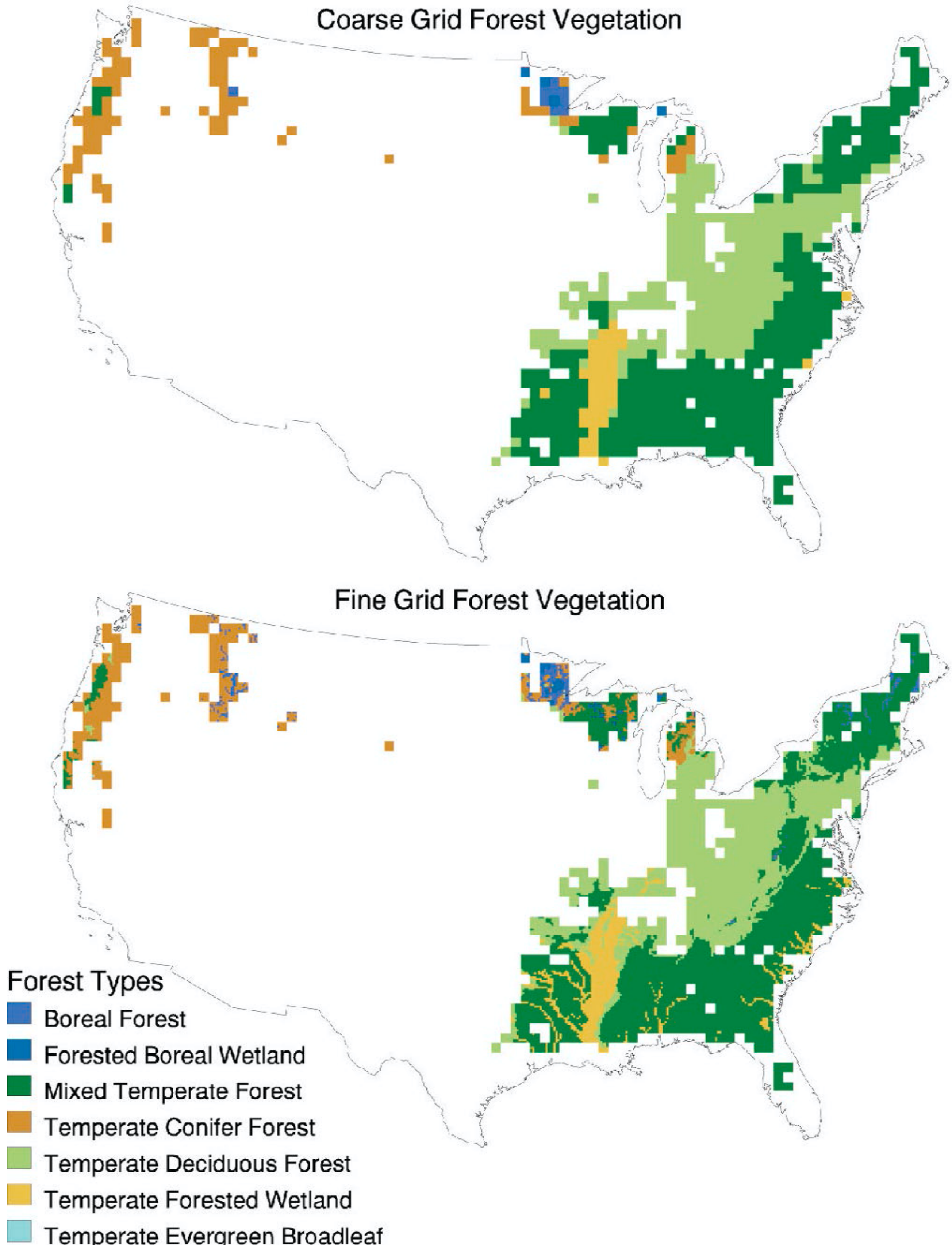

Figure 3.5—Vegetation types mapped at 50 km and 10 km scales.

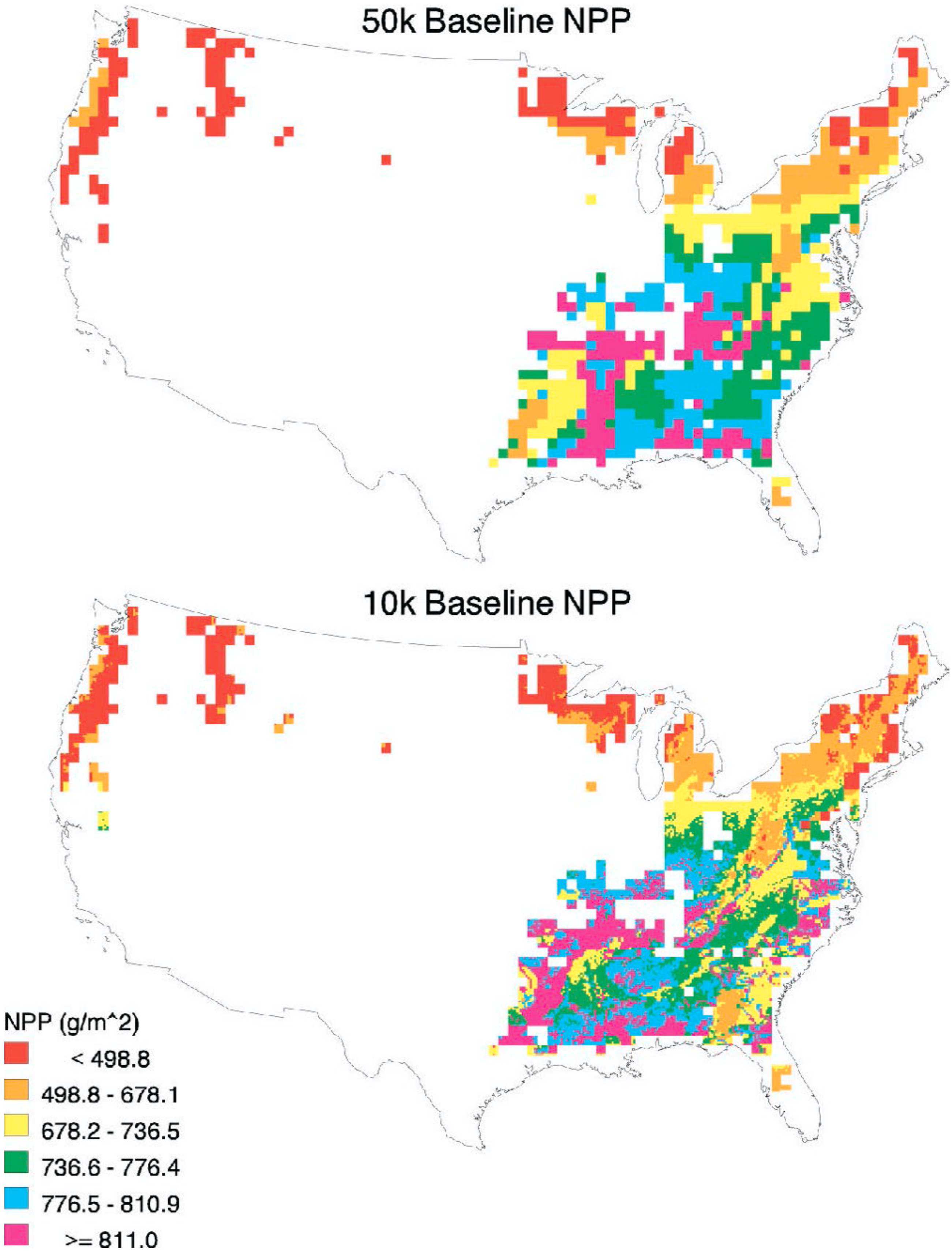

Figure 3.6—Baseline net primary productivity (NPP) at 50 km and 10 km scales.

where d_{OS10}, d_{OS50}, d_{GQ10}, and d_{GQ50} are NPP responses for each fine (d_{-10}) and coarse (d_{-50}) resolution grid to the OSU and the GFDL-Q climate scenarios. This metric is the delta that is passed to the forest sector model and used to modify timber volume increases in the inventory model.

Net primary productivity for all forests increased under both climate change scenarios at both resolutions. Under the OSU climate, NPP of all forests increased approximately 30 percent above the baseline NPP response, whereas under the GFDL-Q climate, NPP of all forests increased less than 13 percent (fig. 3.7). Within each climate change scenario, the response to climate change at the coarse resolution differed by less than 1 percent from the response at the finer resolution. Aggregation error for all forests for the OSU climate (–0.5 percent) was similar to the error for the GFDL-Q climate (–0.8 percent) and both results were similar to the aggregation error of the baseline climate (–0.4 percent). The spatial patterns of these aggregation errors were similar between the two climate scenarios and the baseline across the historic range of temperate forests.

Within forest types, NPP increased from 2.3 to 48.3 percent varying by climate scenario. The NPP response for boreal forests and forested boreal wetlands was 4 to 7 percent greater under the GFDL-Q climate than under the OSU climate. However, for mixed temperate, deciduous, and temperate forested wetland, NPP under the OSU climate was 18 to 40 percent greater than under the GFDL-Q climate. Within each climate scenario, forest NPP responses at the coarse scale differed by less than 2 percent from the NPP response to climate change at the finer resolution for all forest types except conifer, which differed by less than 6 percent. The error in all three climate scenarios was highest in the boreal forests and forested boreal wetlands (11.8 and 10.4 percent, respectively). The absolute value of the aggregation error for the other forests was less than 5.5 percent. Geographically, aggregation error for both OSU and GFDL-Q is concentrated in the same areas as that of baseline climate: around the Great Lakes, in New England, and in the Rocky Mountains. The smallest aggregation error, less than 10 percent, was found in the South and the Mid-Atlantic, an area that had the greatest differences in the NPP response to climate change, –17 to 82 percent.

Relative to the baseline, the percent increases or decreases in NPP are similar across the fine and coarse resolutions within a climate change scenario but differ significantly across scenarios (fig. 3.8). Percent increases in NPP were similar under both climate scenarios in the West, but the responses in the South and Mid-Atlantic forests were dramatically different. For the GFDL-Q climate, the southern and mid-Atlantic forest NPP declined up to 17 percent relative to the baseline NPP whereas under the OSU climate these forests increased in NPP from 10 to 82 percent.

The 50-km grid-cell resolution is most relevant to stand-level forest managers. Aggregation error at this scale is less than 9 percent (two standard deviations), which is within the 20 percent precision that stand-level NPP is measured. Our analyses indicate that aggregation error is the largest in transition regions and in regions with substantial variability in air temperature. Aggregation error is primarily associated with the representation of a mosaic of forest types with a single forest type at 50-km resolution. This source of aggregation error can easily be minimized by making NPP estimates for each forest type within a 50-km grid cell and aggregating estimates based on the proportion of each forest type within the grid cell. This approach has been used by Bonan (1995) as a means of representing vegetative heterogeneity for estimating carbon, water, and energy exchange in the surface boundary layer of general circulation models. Pierce and Running (1995) also found that averaging temperature substantially influenced aggregation error in regions of complex terrain. It may be possible to achieve computational efficiency at 50-km resolution by aggregating temperature for a limited number of elevation bands, making NPP estimates for each elevation band, and aggregating estimates based on the proportion of each elevation band within the grid cell (Nungesser et al. 1999).

The resolution of forest types is most relevant to country-specific economic assessments of the impacts of climate change on timber resources. For example, relative climatic responses of NPP for different forest types in different regions were used as inputs for the last RPA (Joyce et al. 1995). Except for boreal forests and forested boreal wetlands, the mean aggregation error of baseline NPP estimates is less than 2 percent for each forest type. In contrast, aggregation error for boreal forests and forested boreal wetlands is approximately 10 percent. This level of error is caused primarily by the over-representation of boreal and boreal wetland forests and under-representation of more highly productive forests in the 50-km simulation.

In comparison to the baseline simulations, mean aggregation error for the absolute estimates of each forest type in the climate change simulations is similar except for temperate conifer forest (–3.7 percent for OSU and –5.5 percent for GFDL vs. –1.6 percent for baseline climate). Similar to aggregation error, the relative responses of NPP at each resolution are similar except for conifer forests (2.9 and 5.6 percent lower response for the fine resolution OSU and GFDL simulations). The negative aggregation errors and lower responses for temperate conifer forests are associated with the averaging of temperature in the complex terrain in the northern Rocky Mountains and in western Washington, Oregon, and California. Because differences between the responses of NPP at different resolutions are small compared with the responses to different climate scenarios, they could be ignored in impact assessments that evaluate sensitivities to different climate change scenarios.

The national resolution is most relevant to global economic assessments of the impacts of climate change on

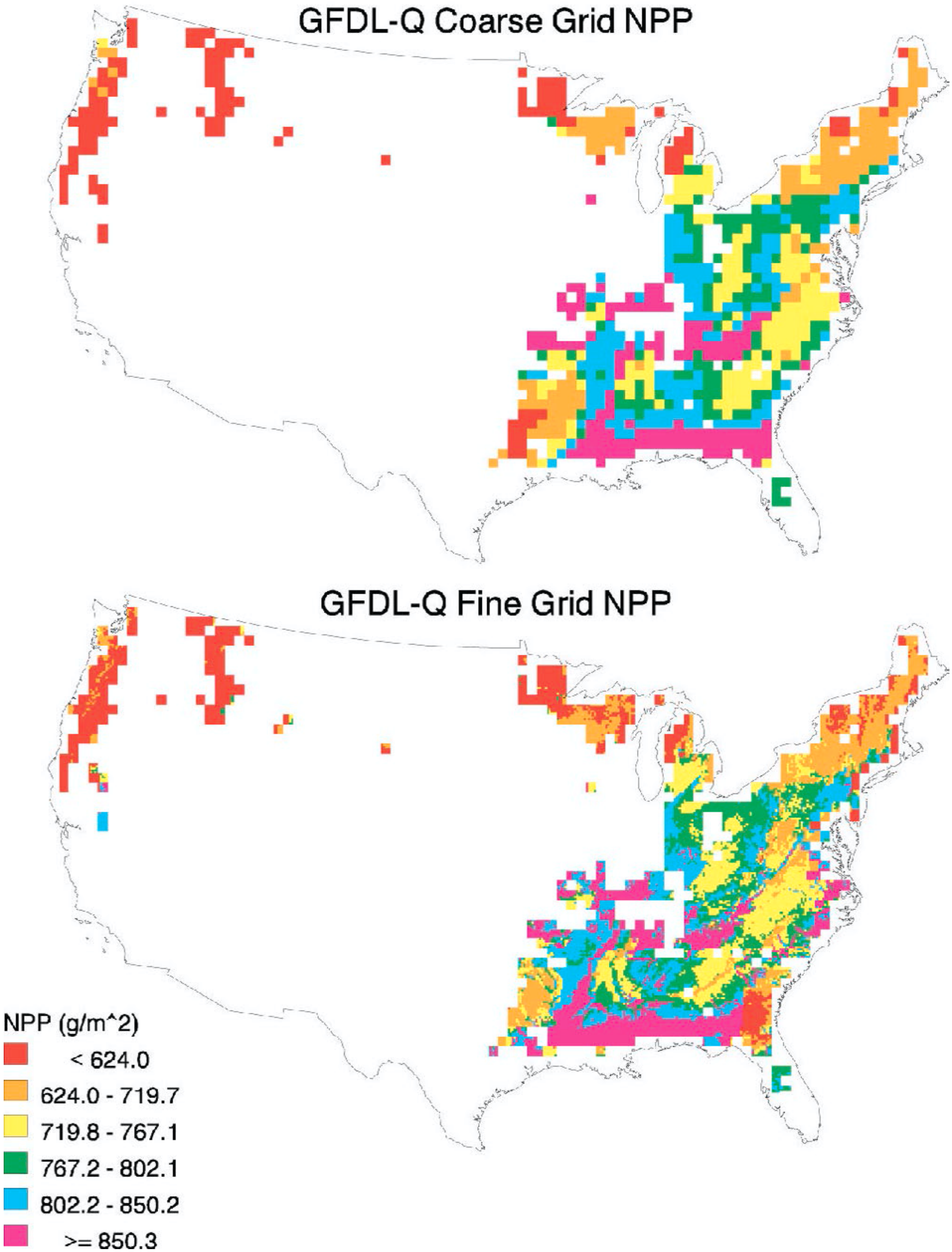

Figure 3.7—Net primary productivity at 50 km and 10 km scales from the GFDL-Q climate scenario.

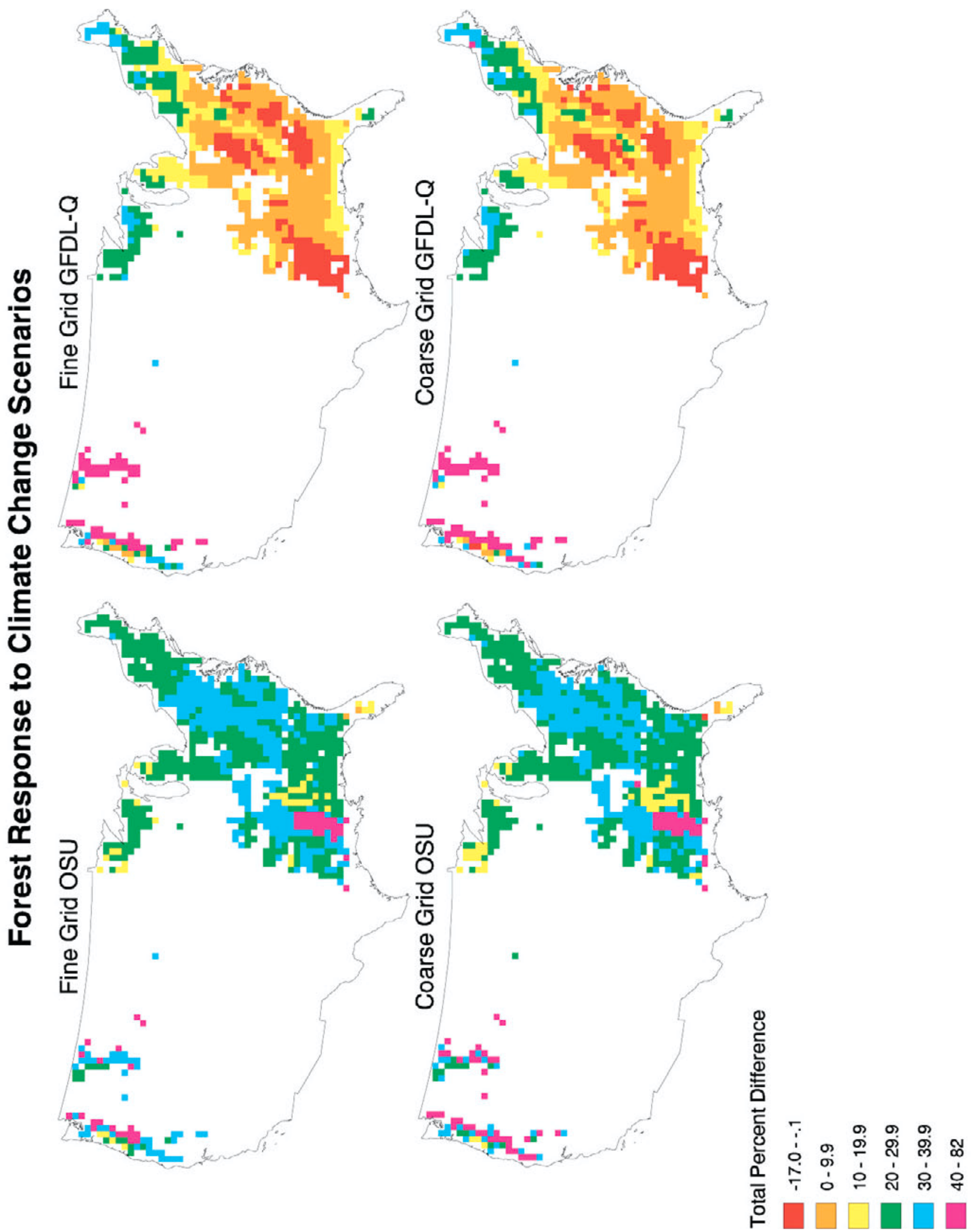

Figure 3.8—Forest responses to two climate change scenarios (OSU, GFDL-Q) at 10 km (fine grid) and 50 km (coarse grid) scales.

timber resources. The relative climatic responses of NPP of hardwoods and softwoods for different regions were used as inputs for a national assessment (Perez-Garcia et al. 1998). For the conterminous United States, aggregation error was 0.4 percent in the baseline simulation, 0.5 percent in the OSU simulation, and 0.8 percent in the GFDL simulation. Responses of NPP across the two spatial scales were 0.3 and 0.7 percent lower for the OSU and GFDL simulations. Because these differences are minuscule in comparison to the national NPP responses, they can be ignored in impact assessments that evaluate sensitivities to different climate change scenarios.

These results indicate that NPP responses of TEM to projected climate change are insensitive to the resolution of inputs, but that aggregation error of absolute NPP estimates is sensitive to the resolution of inputs for some situations. Except for transition areas and regions with substantial temperature variability, these simulations indicate that the use of 0.5° resolution provides an acceptable level of aggregation error at the three scales of analysis in this study.

It is important to recognize that the conclusions in this study are based on two resolutions and one biogeochemistry model. Pierce and Running (1995) used a different biogeochemistry model to simulate NPP for various resolutions ranging from 1 km^2 to 110 km^2 in a region of complex topography. At the coarsest scale, they found coarse-resolution NPP was overestimated by up to 30 percent relative to NPP estimates at the finest resolution. The results of Nungesser et al. (1999) qualitatively agree with those of Pierce and Running (1995).

Because most large-scale biogeochemistry models are parameterized with stand-level data, a systematic analysis of aggregation error with several biogeochemistry models across a range of spatial resolutions from stand to 0.5° (e.g., 100 m^2 to 1 km^2 to 100 km^2 to 2500 km^2) should be undertaken in different forest regions to determine whether our conclusions and those of Pierce and Running (1995) are robust.

Finally, it is important to verify the conclusion about the insensitivity of NPP responses to the resolution of inputs with other biogeochemistry models. By clarifying the scaling issues associated with NPP estimates and responses, these suggested studies would improve impact assessments that rely on the estimates of large-scale ecological models.

Conclusions

Analyses of the impact of climate change on forest productivity, based on experimental research and modeling, would suggest that forest productivity may increase under elevated carbon dioxide, but that the local conditions of moisture stress and nutrient availability will strongly temper any response. Projected increases in productivity from carbon dioxide fertilization appear to be within the same magnitude as potential increases in productivity from timber management treatments. Refinements in the analysis such as analyzing the impact at a finer scale do not appear to alter the results from the last RPA analysis. Incorporating land use changes appears to be a critical next step in the analysis of the impact of climate change on forest productivity.

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Modeling Climate Change Impacts on the Forest Sector

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Introduction

The forest sector has had a relatively long history of applying sectorial models[1] to estimate the effects of atmospheric issues such as acid rain (see Haynes and Adams 1992; Haynes and Kaiser 1990), climate change (Callaway et al. 1994; Joyce et al. 1995; Mills and Haynes 1995; Alig et al. 1997; Perez-Garcia et al. 1997; Sohngen and Mendelsohn 1998), and the forestry impacts of reduced atmospheric ozone (Bently and Horst 1997). The models of the forest sector vary in scope and complexity but share a number of common features and databases. Three aspects in common among them stand out. First, the spatial equilibrium market framework first used by Haynes (1975) and expanded by Adams and Haynes (1980) has provided a structure and framework for many of these efforts. Second, the TAMM/ATLAS model (Adams and Haynes 1980; Mills and Kincaid 1992) itself has provided both relatively complete data sets and estimated relationships for economic processes that are robust and can be aggregated both in market levels and across spatial markets. The third aspect is the rich and unique forest inventory data sets that are available for the United States. This data set designed for assessing timberland conditions and trends provides an essential component to the development of aggregate timber resource models that are integral parts of most forest sector models. It allows for explicit treatments of changes in net growth, land use, and forest type changes.

The purpose of this chapter is to assess broadly our ability to model climate change impacts on the forest sector using two of the three forest sector models that are available in the public domain: the Timber Assessment Market Model (TAMM, Adams and Haynes 1980, 1996)/North America Pulp and Paper Model (NAPAP Ince 1994)/Aggregate Timberland Assessment System (ATLAS Mills and Kincaid 1992)[2] and Forest and Agriculture Sector Optimization Model (FASOM Adams et al. 1996a; Alig et al. 1997). A third model is the CINTRAFOR Global Trade Model (CGTM). It is the application of research started at the International Institute of Applied Systems Analysis (IIASA) and continuing work at the University of Washington (Kallio et al. 1987; Cardellichio et al. 1988, 1989). It has been used by researchers at the University of Washington to examine broad scale global climate change issues (Perez-Garcia et al. 1997).

In addition to the TAMM/ATLAS and FASOM applications that will be discussed in subsequent sections, some of the already listed references represent substantial efforts. For example, Joyce et al. (1995) for North America and Perez-Garcia et al. (1997) for the world examined the effects of climate change on the world's forest products economy using the terrestrial ecosystem model (Melillo et al. 1993) to link ecological change to actual vegetation. The general results show that as primary production increases, timber becomes more abundant, prices fall, and consumption increases. Major timber producers such as the United States and Canada received small positive economic gains from forestry changes. This work also indicates the differences among those particular climate scenarios tend to be small, except in southeast Asia and Oceania for hardwoods, and the northern regions for softwoods. Both studies show that production shifts and changes in trade patterns tend to dampen the effects of climate change in the forest sector. Several studies used variations of the TAMM/ATLAS model structure. For example, Callaway et al. (1994) showed that under a climate of doubled atmospheric carbon dioxide, harvests could be shifted over time, along with changes in tree planting investment, as part of the dynamic adjustment of markets and capital stocks. Haynes et al. (1994) focused on various mitigation strategies that are often suggested for using forests for carbon sequestration. Sohngen and Mendelsohn's (1998) model using a spatial equilibrium structure reduced from some of the relationships in TAMM/ATLAS, focused more on the dynamic adjustment pathway than these earlier studies. They found that markets will mitigate, and even reverse, CO_2 fluxes in contrast to natural response models. Finally, Burton et al. (1997) used a variant of FASOM to look at three scenarios of extreme growth-rate change induced by global climate change and found that impacts are felt more strongly by producers than consumers, and more by southern producers than producers in other regions.

Several unique gaps or opportunities are apparent from this array of applications of different approaches to forest sector modeling. First, most of these systems depend on forest inventory/resource models to provide exogenous variables in stumpage supply relations. These resource models are invariably based on actual (or current) forest vegetation types and extents that reflect exten-

[1] *A forest sector model, in general, combines activities related to the use of wood: forest growth and harvest; the manufacture of pulp, paper, and solid wood products; and international trade and intermediate and final consumption of these products (Kallio et al. 1987).*

[2] *Hereafter called TAMM/ATLAS.*

sive human modifications. Integrating economic and ecological models requires some means to relate actual vegetation to the projected changes in potential vegetation that result from process-based ecological models. Several of the applications above used the terrestrial ecosystem model (Melillio et al. 1993) as one such linking device. Second, most of the models described above have not addressed in detail the reallocation of land between forest and agricultural sectors if productivity is impacted by global climate change. Landowners continually consider shifts in land use between agricultural crops and forest, and shifts within forestland among different intensities of land management, some of which involve shifting from lower value to higher value tree species. In addition, there is the concern about land conversion to developed uses. A third set of opportunities involve consideration of the propensity for persistent change in the levels and types of human demands for various forest goods and services.

Comparison of Model Structures

The Timber Assessment Market Model (TAMM)

The TAMM system is one of the best known examples of what are termed forest sector models. Since its inception in the late 1970s, this system of models has undergone a number of extensions and revisions designed to improve the realism of its projections and the utility of its output to resource analysts and policy makers. Details about TAMM projections and underlying assumptions are described in detail in the 1989 USDA Forest Service RPA Timber Assessment (Haynes 1990) and 1993 RPA Timber Assessment Update (Haynes et al. 1995). TAMM is a bioeconomic model that provides an integrated structure for considering the behavior of regional prices, consumption, and production in both stumpage and solidwood product markets and incorporates a bilaterally linked model of timber resources and timber supply. In its current form it includes the North American Pulp and Paper Model (NAPAP) for paper products and ATLAS to model timber resources.

NAPAP (see Ince 1994) uses linear programming to solve for market equilibrium in spatially specific markets. It includes regional supply functions for pulpwood and recovered paper (recycling), and a detailed representation of production capacity and supply for all principal grades of market pulp, paper, and paperboard, in five North American production regions. The model also includes demand functions for all end products, with separate demand functions for U.S. domestic demand, Canadian domestic demand, and demand from various trading regions for export from the United States and Canada.

ATLAS (Mills and Kincaid 1992) is used as an inventory projection system and estimates of available timber inventory are used in the timber supply relations for each region and owner. ATLAS was developed to model timber inventories at subregional, regional, and national scales using timberland inventory data collected by the various USDA Forest Service Forest Inventory and Analysis Units (FIA). The data are stratified and aggregated by species group (forest type), site productivity, and expected management class. Growth and yield models are estimated representing a broad mix of conditions and management intentions. In each simulation period, inventory change is the result of growth, area change, and harvest. The area adjustment is derived from projections by area models (see Alig 1985; Alig et al. 1990).

Forest and Agriculture Sector Optimization Model (FASOM)

The Forest and Agriculture Sector Optimization Model (FASOM) was originally developed for the U.S. Environmental Protection Agency to estimate the market impacts of carbon sequestration options for both the agriculture and forest sectors (Alig et al. 1997). Unlike TAMM, FASOM is a price-endogenous quasi spatial multi-period equilibrium model. Its objective function maximizes the discounted economic welfare of producers' and consumers' surpluses in the U.S. agriculture and forest sectors over a finite time horizon. FASOM operates on a decadal time step, with projections made for 10 decades; however, policy analysis is limited to results for the 50 year period from 1990 to 2040. FASOM employs a single national demand region for forest products, which treats only the log market portion of the sector. The nine U.S. timber supply regions are similar to TAMM regions, except for combining the Northern and Southern Rocky Mountain regions and separating the "Corn Belt" and "Lake States" portions of the North Central region because of their agricultural importance. Private timberland in FASOM represents a reaggregation of the ATLAS model; the strata are differentiated by: 1) class of ownership (forest industry and nonindustrial); 2) forest type (four classes describing species composition, either softwoods or hardwoods, in the current and preceding rotation); 3) site productivity (three levels of potential for wood volume growth); 4) management intensity (four discrete timber management regimes); 5) suitability for transfer to or from agricultural use (four land suitability classes for crop or pasture plus a "forest only" class that cannot shift use); and 6) 10-year age class (ten).

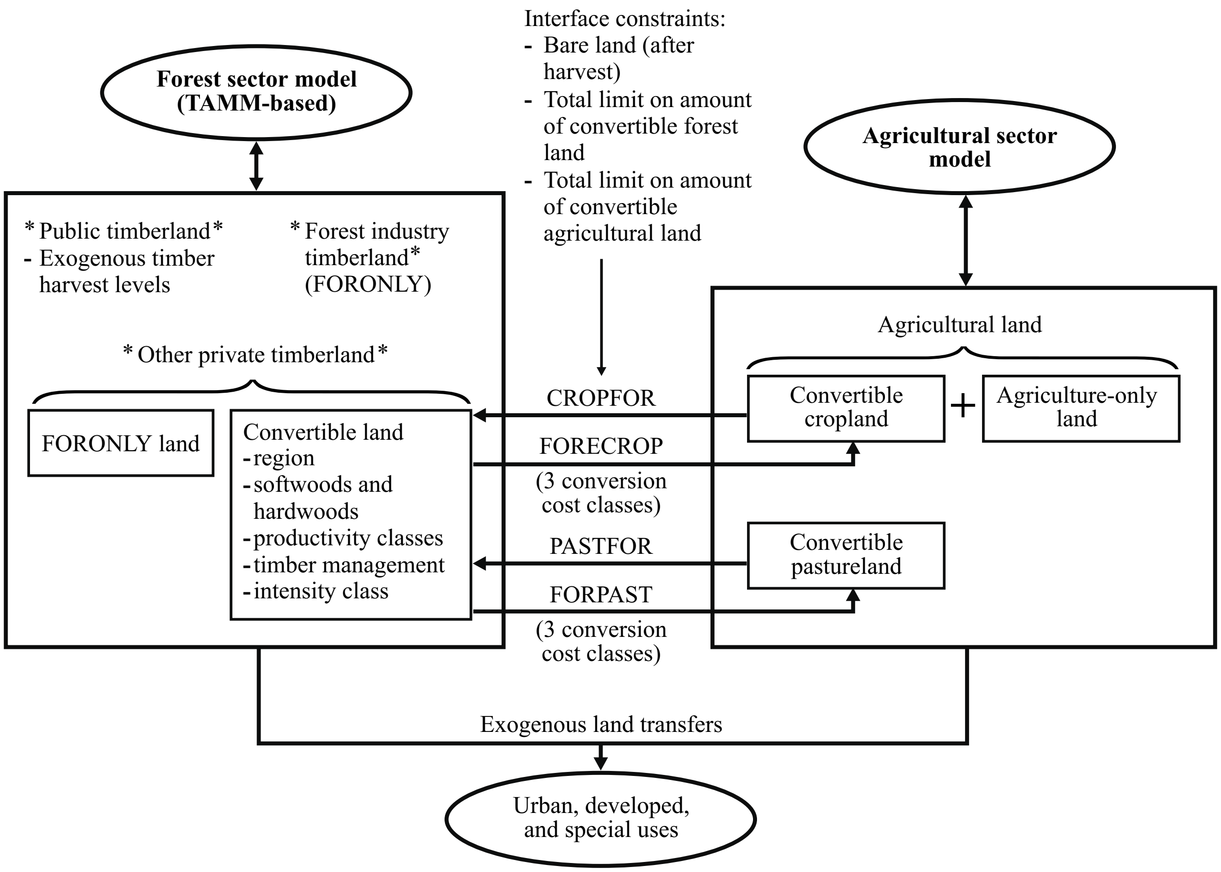

Figure 4.1—Linkage of forestry and agriculture sectors in FASOM.

Endogenous variables include: 1) timber harvests and log prices for nine U.S. regions, two species groups, and three classes of products (sawtimber, pulpwood, and fuelwood); 2) timber management investment activity for two private owner groups (forest industry and other private); 3) agricultural prices and production in 11 regions for 50 primary and 56 secondary commodities; and 4) the amounts of land used in, and transferred between, the two sectors. All exogenous forestry elements of the model are held constant after the fifth decade. The model values terminal inventories (at the end of the finite projection period) in both sectors assuming perpetual, steady state management following the terminal year of the explicit time horizon (Adams et al. 1996a).

The agriculture sector in FASOM is adapted from the Agricultural Sector Model (ASM)[3], aggregated to regions matching those in the forestry sector. ASM is a spatial price-endogenous agricultural sector model (Chang et al. 1992), with constant elasticity curves used to represent domestic consumption and export demands as well as input and import supplies. ASM was originally constructed as an essentially timeless, long-run equilibrium model. To link ASM with the decade cycles in FASOM and the forest sector, where market interventions may take several decades to play out, the model was converted to an annual format. Updating between decades was accomplished using projected growth rates in crop yields, domestic demand, exports, imports, and cropland availability.

One real strength of FASOM is the links between land inventories in the agricultural and forest sectors (fig. 4.1). Suitable land can move, at any time, between agricultural and forest uses, based on considerations of inter-temporal profitability and subject to the availability of resources and the specific provisions of particular policies.[4] The planning problem simulated in FASOM allows landowners to foresee the profitability consequences of all the pos-

[3] The Agricultural Sector Model (ASM) is described by McCarl, B. C.; Chang, J. Atwood; Nayda. W. in "The U.S. Agriculture Sector Model." On file with the Social and Economics Values Program, Forestry Sciences Laboratory, Pacific Northwest Station, 3200 S.W. Jefferson Way, Corvallis, OR 97331.

[4] Rising relative prices for urban and developed uses, at the top of the economic hierarchy of land use, prompt exogenous shifts of forest and agricultural land to urban/developed uses, by region each period, along with some timberland reclassified to reserved uses (Alig et al. 1990; Alig and Wear 1992).

Table 4.1—Component parts of contemporary forest sector models.

Model component	Type of variable	Process model/function
Product demand	Prices, macro economic variables	New product diffusion
Product supply	Costs, capacity levels	Capacity adjustment
Stumpage demand	Conversion factors	Materials balance relationships
Stumpage supply	Prices, inventory attributes	Resource projection system (ATLAS)
Land use changes	Land prices, land types	Area-change projection system

sible agricultural and forest uses of their land over time. Through the land type classes, hectares of nonindustrial private timberland that could be converted to cropland and pastureland and also agricultural land that could be shifted into forestry can be identified. Estimates of the area of convertible forestland are from USDA estimates of forestland with medium or high potential for conversion to crop or pasture use; area estimates for convertible agricultural land are drawn from Moulton and Richards' (1990) study of land suitable for tree planting.

TAMM and FASOM Comparison

The TAMM and FASOM models complement each other, and they are related in several important ways (Alig and Adams 1996). Both models contain representations for the various component parts of contemporary forest sector models (see table 4.1). As discussed in this chapter, these representations vary between the two models. Externally, the models have some common links, but internally the models offer different solution mechanisms. In brief, both models embody the four key components of timber supply modeling systems identified by Alig et al. (1984): land allocation, growth and yield projections, harvest flows, and forest management investment.

In terms of some of the model components illustrated in table 4.1 there are differences between TAMM and FASOM. TAMM deals relatively explicitly with the first four model components by treating land use changes as an exogenous process. FASOM focuses on the stumpage market (especially the last two components), relying on TAMM/ATLAS for product market detail that is collapsed to a set of derived demand relationships for logs aggregated at the national level. In the case of both models, the approach to estimating demand considers demographic and economic variables, including population growth, housing starts, household formation and size, and technology improvements—all of which are derived from other models and economic forecasts.

An important distinction is the solution algorithm used to simulate market behavior. The solution of TAMM represents a spatial equilibrium in the markets modeled for each year of the projection period. A spatial equilibrium model solves for equilibrium between price and quantity simultaneously in multiple spatially distinct markets at two different market levels. These solutions represent production, consumption, and price time paths that are estimates of outcomes of contemporaneous interactions in freely competitive markets. FASOM solves for equilibrium in the stumpage/log market recognizing spatially distinct timber supply regions. It differs from TAMM in that it also solves for the intertemporal equilibrium. That is, landowners make investment decisions guided by exact knowledge of future prices and harvest levels.

In terms of the resource situation, both TAMM and FASOM share the same base inventory and provide explicit treatment of actual vegetation attributes for 145 million hectares of U.S. private timberlands. Using actual types facilitates analyzing the combination of bio-physical, ecological, and socioeconomic forces that influence the amount of land allocated to major land uses and forest cover types in the United States.

In both models the timberland base is adjusted over time for the movement of land between forest (timber production) and non-forest (including, agricultural, urban, and reserved) uses. In TAMM, land allocation is exogenous, provided by models of land use changes that assume landowners are present value (quasi-rent) maximizers in allocating land to alternative uses (e.g., Alig 1985). Examples are usually cast in the context of two primary competing uses, but in most cases methods can be readily extended to multiple uses. Systems of equations describe the major land uses (Alig 1986). In FASOM this forestry-agriculture land use margin is endogenous. In both models, when timberland shifts to a non-forest use a portion of the timber volume is often harvested and counted in the current aggregate cut from the stratum. This reflects the process of land clearing or volume reduction associated with most land use changes in the private sector (Alig et al. 1990).

The investments are associated with various timber management practices. The yield regimes derived for the ATLAS model are used in both models. Some of these are empirically based (based in part on the regional FIA survey plots, Powell et al. 1993), while others are products of specific stand models. Assignment of areas to management intensity classes (MICs) are based on data derived from field measurements and judgments of USDA Forest Service inventory analysts and experts in industry and other groups. A MIC is defined by a combination of silvicultural activities including, but not limited to, improved regeneration, stocking control, commercial thinning, and fertilization. Land can shift among MICs over time to reflect changes in timber management investment. This investment is treated as exogenous in TAMM/ATLAS, where a schedule of management treatments is developed for all private lands considered in the ATLAS projection. This schedule is based on current expectations but is not sensitive to endogenous price changes or projected market elements (though it may be changed through considerations of model outcomes). In FASOM, the investment actions are part of the solution. The extent and timing of the MIC shifts represent an optimal solution based on the "perfect knowledge" of future markets.

Results

Both TAMM and FASOM have been used in the context of scenario planning to examine a wide range of alternative scenarios related to the role of forestry in climate change. In this section we describe generalized results for several of these scenarios. TAMM was used to examine the effect of increased atmospheric carbon dioxide on the timber situation in the United States and it has also been used to examine the use of various forest policies to mitigate global climate change, but that work is discussed elsewhere (see Chapter 8, and Haynes et al. 1994).

FASOM has been used to examine several alternative futures of increased forest carbon sequestration. The FASOM results about mitigation options are summarized in Chapter 8.

Climate Change and Forest Productivity—TAMM

TAMM/ATLAS has been run for scenarios where forest productivity was sensitive to climate change (for details see Joyce 1995). The nature of the prospective climate change was a doubling of atmospheric carbon dioxide by the year 2065. A biogeochemistry model (TEM) was used in conjunction with general circulation models (GCMs) to project changes in forest net primary productivity (NPP). A key assumption was that changes in annual NPP of potential vegetation were proportional to annual changes in the rate of projected forest growth. These changes were applied to the forest types in ATLAS, effectively linking a biogeochemistry model to a forest sector model.

Results from the baseline projections of the 1993 RPA Timber Assessment (Haynes et al. 1995) were compared with three scenarios depicting a maximum, mean, and minimum set of changes in NPP. With the exception of a negative impact on Southern hardwoods in the minimum projections, the change in NPP was positive for all forest types in all regions. In total, the projected increases in NPP far outweighed declines. This set of scenarios increased forest growth over all regions, which led to a 3 and 22 percent increase in inventory on private lands (13 to 112 billion cubic feet above the base). Where the RPA baseline projections showed declines, the climate change scenarios show increases.

One fundamental result from the TAMM/ATLAS simulations has been to show how increases in growth eventually impacts levels of harvest. Growth accumulates as inventory volume, leading to increases in the timber available for harvest; there is a market price response; and there is an increase in the harvest. But not only is this increase in harvest a lagged response, the magnitude of the increase indicates a relatively small consumer response to lower lumber prices. Demand for solidwood products is derived from consumption of houses, other types of buildings, and a wide range of consumer and industrial products. Demand for paper is primarily influenced by overall economic growth. In both cases downward changes in wood prices (or fiber in the case of paper) represent only a small proportion of total production costs. This is a significant result and it is consistent with work referenced earlier that studied the market impacts of acid rain and atmospheric ozone. These applications illustrated two important features of TAMM: first, the explicit temporal structure detailed the relatively long lag between changes in net growth and eventual changes in harvests and attendant economic impacts; and second, TAMM's explicit treatment of both stumpage and product markets allows for empirical estimation of producer and consumer surplus measures that follow the usual economic conventions (for both product and factor markets).

TAMM/ATLAS has also helped shape perceptions that with a change in available supplies of harvestable inventories, there is a shift of harvest both among and within regions as product production adjusts to take advantage of lower cost raw materials in some regions. Shifts occur between fiber types and between ownership classes. Under the TEM scenarios there is a higher overall supply of softwoods and lower prices. Harvest then shifts toward the industry ownership. In most regions, capacity increases and the harvest expands faster on industry land than it does on nonindustrial timberlands. In the Pacific Coast, the nonindustrial harvest declines under all three scenarios. Many of these changes are stimulated by changes in stumpage prices, which relative to softwood sawtimber stumpage prices in the RPA base projection (that follow an upward sloping path to 2040), these new projections show a leveling of prices by 2015 and then prices begin to decline and by 2040 drop below levels predicted for 1995.

This past work also showed the importance of trade, in that without a change in market demand or a change in Canadian inventories, domestic harvest replaces Canadian harvest. Imports of Canadian lumber decrease both in total volume and as a percent of the market. Cheaper U.S. raw material has both increased lumber consumption and made the Canadian lumber less competitive. Without explicitly modeling Canada's resource sector, it is difficult to say how a change in Canadian productivity would affect the level of U.S. imports.

Climate Change and Forest Productivity—FASOM

The FASOM framework has been used as a platform for investigating implications of increased forest carbon sequestration (Alig et al. 1997, 1998; Adams et al. 1999), constraints on available funds for forest investment by private owners (Alig et al. 1999; Adams et al. 1998), biomass analyses and natural resource policies (Alig et al. 1997). Investigating the sensitivity of FASOM projections to a range of different assumptions offers a unique perspective, particularly where policy makers are concerned with linkages between forestry and agriculture and with both economic and environmental consequences of different policy alternatives (e.g., information on future non-timber resource conditions, such as wildlife habitat, Alig et al. 1998).

The FASOM model was applied to examine the dimensions of economic impacts due to hypothetical biological responses to global climate change (Burton et al. 1995). This exploratory study considered eight possible scenarios for global climate change effects. They were designed as an attempt to dimension the potential impact of climate change. The first four climate/biological response scenarios evaluate the effects of across-the-board changes in tree growth rates, or yield, for each decade. The first scenario postulates an increase of 5 percent in tree growth rates everywhere in the United States. The second postulates an across-the-board decrease in growth rates of 5 percent. Two scenarios consider national growth rate changes of plus 10 percent and minus 10 percent.

In addition, two scenarios explore the different effects from increased warming at different latitudes in the United States. The warming, coupled with a slight decline in precipitation, may negatively impact timber yields in the southern United States. At the same time, yields in the northern United States may rise. Therefore, a pair of southern decline scenarios were constructed. One postulates a 5 percent decline in yield in the South, a 5 percent increase in the North, and no change in other regions. A second southern decline scenario explores the impact of a 10 percent decrease in the South, a 10-percent increase in the North, and no change in other regions. The baseline case is similar to that described by Haynes et al. (1995) for the 1993 RPA Assessment Update.

McCarl et al. (2000) have since developed response functions that represent a wide range of scenarios for the biological response of forests to climate change, ranging from small to large changes in forest growth rates. The response functions are used to characterize broad impacts of climate change on the forest sector. Aggregate impacts (across all consumers and producers in society) are relatively small but that producers income and future welfare 30–40 years in the future are most at risk.

The FASOM model projected characteristics of forest products production, price levels, timber management changes, land transfers, and economic welfare effects.[5] The results of the exploratory study indicate that market responses to climate change will vary by region. If yields increase nationally, the North can produce relatively more forest products. If yields decrease nationally, the South can produce relatively more. If yields in the South decrease, while they rise in the North, production of forest products is projected to shift away from the South. If stand establishment costs rise in the South, production may also shift away from the South.

Changes in timber producer and consumer prices under the climate change scenarios relative to the base scenario are fairly small in magnitude. When southern costs of timber production rise, timber price levels increase more substantially than in other scenarios.

The economic welfare impact of the global climate change scenarios is small—less than a 1 percent change from the base scenario across all eight hypothetical cases. In general, when yields rise, consumers gain. When yields fall or costs rise, producers gain. In FASOM any change in future conditions is optimally anticipated (from a net social welfare viewpoint) and investment is freely flexible to vary over time. A representation of "real world" behavior would doubtless be somewhat less adaptable, recognizing limitations of the decision maker. The structure of the present model has been modified to examine some of these questions of "stickiness" in product and capital markets (Alig et al. 1999; Adams et al. 1998), including limits on investment borrowing or capital budgets, increasing marginal costs of borrowing, and uncertainty regarding future market conditions. The FASOM results in the investments case are closer to those projected by the TAMM model for comparable scenarios. Use of both the TAMM and FASOM models reveals the differences between: 1) likely future paths for the forest sector if historical relationships between key variables continue; versus 2) production possibilities and optimal responses to external events (e.g., climate change) and policies. FASOM assumes perfect foresight and optimal adjust-

[5] *Global change mitigation analyses using the FASOM model are discussed in chapter 8 of this report.*

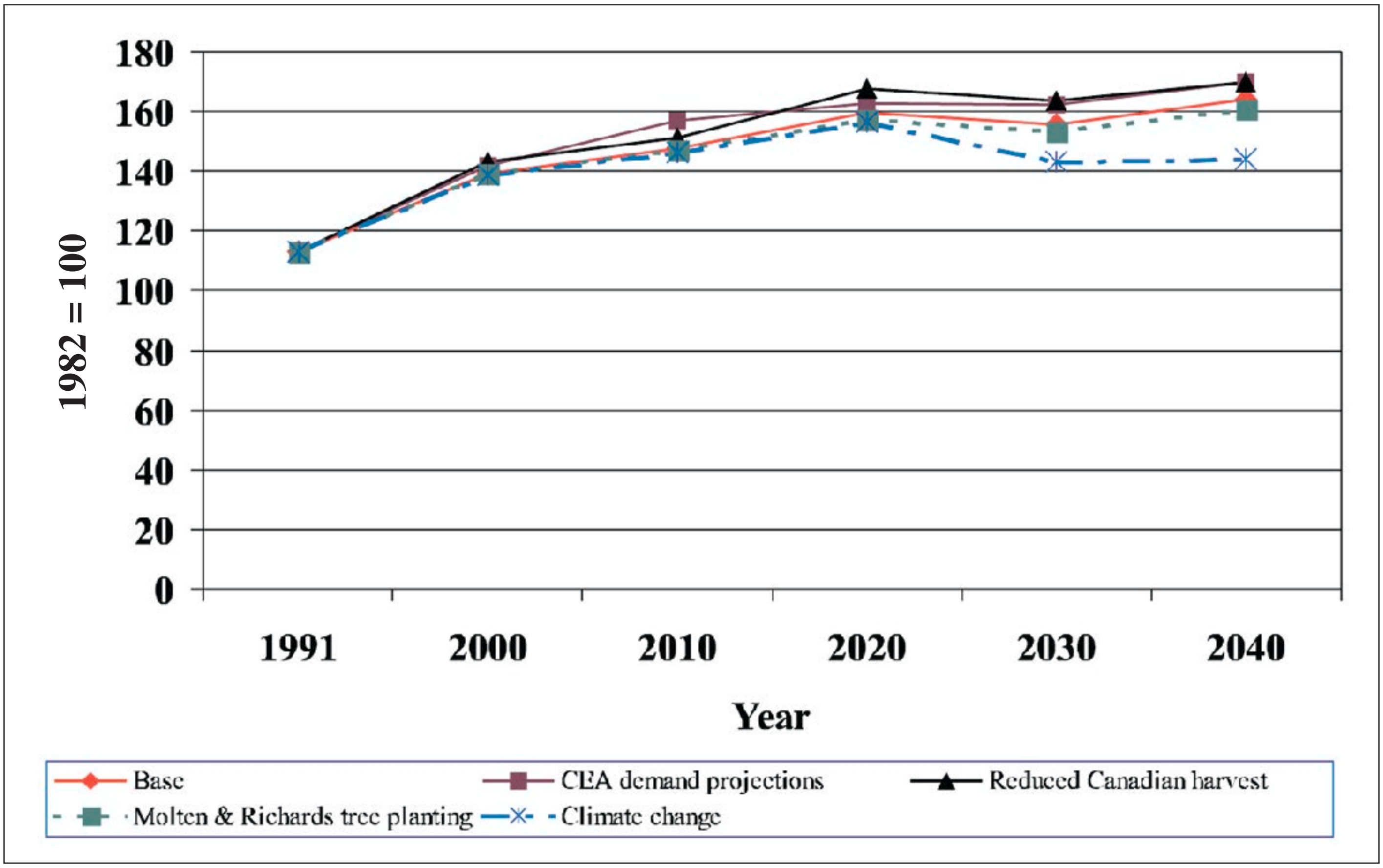

Figure 4.2—Softwood lumber price index under different views of the future including one prospective view of climate change. (The CEA projection represents the results from an alternative macro economic forecast developed by the Council of Economic Advisers in 1994.) Source: Haynes et al. 1995.

ments in the unfettered case. As such it shows a greater shift to pine plantations, and as timber prices fall, an increase in land moving from forest to agriculture and a decrease of investment in pine plantations.

With FASOM having all four key timber supply modeling elements as endogenous components, a different set of projected adjustments (both temporally and across regions) are possible than with TAMM. A range of adjustments is discussed in Chapter 8, where analyses of global change mitigation strategies using the FASOM model are reviewed.

Model Uncertainties

These two forest sector models are useful in developing a portfolio of possible impacts of human uses (in a commodity sense) on forests under climate change. Such factors are not normally considered with most climate change simulation models. They show how and the extent that climate-change influences on U.S. timberlands will be mitigated by market feedbacks between the natural resource base and the production and consumption of forest products. This helps place the issue in context. As shown in figure 4.2, the prospective impacts on the U.S. forest sector of one view of global climate change is overshadowed in the near term by other contemporary policy concerns such as habitat conservation strategies that involve reductions of timber harvest on federal timberlands in the west (e.g. Adams et al. 1996b). This raises questions about the timing of the often cited prospective catastrophic ecological declines associated with climate change and the accompanying specter of economic dislocation within the U.S. forest sector. This scenario needs to be examined in light of the extent and speed of changes induced by price signals from timber markets. That is, to what extent will the expectations of lower prices in the future associated with climate-change reduce land management actions taken in the next decade?

Like all views of the future, those discussed in this chapter are highly dependent on the underlying assumptions, including model form. An ideal system can vary with the policy analysis needs, but a truly ideal system is not possible because of limited resources and data gaps. Projections of timber markets require assumptions concerning future: 1) product demand, which is largely based on projected population and employment; 2) capacity, which is dependent on flexibility in location of production and profitability; and 3) available timber inventory, which is dependent on area in timberland, and minimum standards for tree size (age) by owner and fiber type. Challenges also arise related to the scale of the analysis. For example, the timber inventory modeling assumes a broad range of habitats and species can be aggregated into forest types across large regions. At this level the changes in inventory did not recognize the potential responses that might be associated with individuals in the system or how adaptive forest management regimes might affect those individuals. Assumptions are made in averaging over the range of variability in ecological relationships associated with temperature changes, rainfall patterns, nutrient cycling, and thresholds in growth or site carrying capacity related to the ability of ecosystems to adapt to change. Additionally, changing vegetation patterns are assumed not to influence climatic conditions.

Though these models accounted for harvest and importation of wood products from Canada, they did not account for climate induced changes in Canadian inventories. Melillo et al. (1993) found that the higher latitude forests experienced increases in productivity at least equal to those of the Northern United States. This result would likely lead to a possibly higher import level than previously considered.

Other assumptions need to address the context surrounding forest sector issues. For example, worldwide assessments of human influence should incorporate differences in likely use of forest resources. Specifically, developing countries tend to view forests as a source of food and fuel whereas in North America forests are viewed as a source of industrial wood products and a range of amenities.

Finally, U.S. product markets have grown 72 percent (1.4 percent per year) over the past four decades while at the same time forest resources have grown 28 percent. In the next five decades we expect slowing in the growth of consumption and in forest resources (Haynes et al. 1995). Prospective changes in prices signal changes in tastes, industry location, and incentives to landowners, all of which act to mitigate potential impacts of climate change. Two important concepts to consider when developing assessments of forests with respect to climate change are: 1) include effects of humans as the most adaptable component of the system; and 2) make clear distinctions between science and policy (or politics).

Ongoing Work

There have also been efforts to broaden the examination of the timber sector under climate change by linking TAMM with a biogeography model known as Mapped Atmosphere-Plant-Soil System (MAPSS; Neilson 1995). (See Chapter 2 for a more comprehensive description of biogeography models and MAPSS.) To date, we took a linkage approach opposite that used with TEM. The ATLAS timberland inventories were translated and then reaggregated into the physiognomic vegetation types projected by MAPSS. This allowed ATLAS to simulate simultaneous changes in cover type and forest productivity in terms of MAPSS.

Difficulties arose when we attempted to match the RPA base projection in terms of the physiognomic cover types. The MAPSS types were broader, including multiple forest types. The type transition and land area shifts already in the base projection had to be preserved so as not to lose the growth and harvest interaction at finer scales. Further difficulty arose when ATLAS could not match the hardwood and softwood aggregation required by TAMM. The new vegetation types cut across the traditional softwood and hardwood categories in which the forest types fit. Time and expertise have not been available to reprogram the TAMM/ATLAS model linkages. It is assumed that in all likelihood, large shifts in vegetation and productivity projected under some scenarios would require significant recalibration of the economic side of the model. The project remains a significant research challenge for the future.

Conclusions

Both TAMM and FASOM are affiliated models that share many common features and present similar views of the relationship between forests and atmospheric issues. Both help to place prospective atmospheric issues, concerns about ecological change associated with climate change, and concerns about the effectiveness of various mitigation measures in context. Both models challenge ecologists and policy analysts to be explicit in the size, location, and timing of various impacts and to consider the transition from current vegetation and to gauge the tradeoffs between near term policy concerns and long-term ecological impacts. While not explicitly addressed elsewhere, both models offer a common framework for integrating biophysical and social systems and for tracing how changes in typically biophysical attributes (growth, area of certain types, etc.) affect various measures of economic benefits and costs. In that role these models operate at the interface of science and policy where the emphasis is on how models themselves improve the information available for decision makers. That same information from the policy perspective helps shape perceptions about the effectiveness of various management actions.

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Carbon Sequestration in Wood and Paper Products

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Introduction

Recognition that increasing levels of CO_2 in the atmosphere will affect the global climate has spurred research into reducing global carbon emissions and increasing carbon sequestration. The main nonhuman sources of atmospheric CO_2 are animal respiration and decay of biomass (U.S. Congress OTA 1991). However, increases in atmospheric levels are attributed mainly to fossil fuel burning and land use change. While efforts to hold down emissions of CO_2 continue, increases in CO_2 emissions can also be offset, to a degree, by accumulation in carbon sinks such as plant biomass and oceans. It is therefore prudent to focus research efforts both on increasing carbon in sinks and reducing carbon emissions.

In 1990, U.S. CO_2 emissions were 1,367 Tg carbon equivalent (Clinton and Gore 1993), where Tg is 1 million metric tons. Wood and paper products play an important role in mitigating these emissions by sequestering carbon. There are currently large stocks of carbon in forests, in wood and paper products in use, and in dumps and landfills. The size of these carbon stocks is increasing. In 1990 approximately 145 Tg of carbon, or 10.6 percent of the level of U.S. emissions was harvested and removed from forests for products. If a substantial portion of this carbon could be prevented from returning to the atmosphere, it could be a notable contribution to mitigating carbon buildup in the atmosphere.

We use the term sequestration to refer to the net sequestration, over a period of time, in a stock of carbon: carbon in forests, carbon in forest products in use (including net imports), or carbon in forest products in landfills. This expands the use of the term beyond its common use referring to net sequestration of carbon to forests.

Carbon sequestration to wood and paper products has been assessed in several other studies. Some studies assess carbon sequestration for a range of hypothetical conditions of forest growth, harvest, end use, and disposal (Schlamadinger and Marland 1996). A worldwide study by Winjum et al. (1998) estimates net flows of carbon out of forests and into products using the two accounting frameworks used in this study—the stock change method, and the atmospheric flow method. They use simplified assumptions to make estimates of net stock changes, and net emissions to the atmosphere by world region and for selected countries. They include estimates of logging residue and assumed decay. Their results, as noted below, are close to ours even though their methods are very different. Other studies focusing on the United States, similar to this one, estimate the actual stocks and flows of carbon from U.S. forests to products in use, to dumps or landfills, and to burning and emissions from decay including reconstruction of historical flows and projections (Heath et al. 1996; Row and Phelps 1996). This study presents similar results with three improvements: 1) use of greater detail in the changing composition of end uses of wood and paper products; 2) inclusion of net imports of wood and paper products in carbon sequestration estimates; and 3) use of new, much lower decay estimates for wood and paper in landfills including separate estimates of CO_2 and CH_4 portions. These improvements help provide a clearer understanding of how sequestration to products may change.

Our purpose is to show an in-depth method of providing historical estimates and projections of U.S. carbon sequestration to wood and paper products. We compare those estimates to amounts sequestered in U.S. forests (an estimate of carbon stock change in the United States). We also show how amounts used to estimate the net sequestration in products and forests each year may be used to estimate the net removals of carbon from the atmosphere to the United States each year (an estimate of carbon flow to the United States). We discuss how patterns of wood use have changed and will change and how they will influence the pattern and amounts of carbon sequestered.

Methods

Historical data and long-range projections were used to track roundwood and carbon disposition through to end uses such as housing or paper. To track carbon beyond end uses to waste products, we estimated burning, disposal, and decay for waste generated in the process of using primary products, and for rates of product disposal from end uses, decay, and burning.

The scope of the analysis is focused and limited in a number of ways. We track carbon harvested and removed from roundwood harvest sites. The decay and carbon emissions from logging residue is contained in separate estimates we display of net sequestration in forests. We did not estimate the amounts of carbon released due to fossil fuels burned by harvesting equipment, or to power primary or secondary wood and paper products mills, or to make final products using wood (such as housing). We note how much carbon wood and paper products sequester that may offset such emissions. We show estimates of net sequestration in forests; the estimates include all carbon accumulation in trees and soil and all deductions for decay of dead trees including logging residue, and deductions due to emissions from forest fires. We include emissions from all burning of wood residue and discarded wood for energy or incineration and, over time, all regrowth of all trees. Our model projections do not include projections of biomass plantations for energy production. We did not calculate if, over time, the degree to which the effect of harvesting and using wood for fuel increases the growth in forests over the growth that would occur without wood burning, so as to reabsorb the carbon emitted by burning. This is an important question for further research. We estimated the amount of harvested and used carbon and its disposition starting in 1910.

For our historical estimates (post-1909) and projections, we tracked carbon added to, and emitted from, stocks of wood and paper products in the United States. Net sequestration to U.S. carbon sinks come from wood in trees harvested in the United States and from net imports (imports minus exports) of logs and wood and paper products. Historical harvest and product use data are needed to estimate future emissions from products that were manufactured in the past. Carbon contained in harvested timber and net imports is tracked through primary processing into products and end uses (fig. 5.1) (adapted from Row and Phelps 1996). Wood or paper residues are generated at all phases of processing and are either reused in a product, burned with or without energy, or dumped (historically) or landfilled (currently). Wood and paper products are tracked to various end uses, where they have a limited life span and are retired from use and sent to landfills or burned. The fate of logging residues were not considered in this model, since decay and emissions from these residues are modeled as part of the forest ecosystem and included in estimates of change in carbon sequestered in forests (Heath and Birdsey 1993; Birdsey and Heath 1995).

Historical data on wood harvest and end use from 1910 through 1986 are from USDA Forest Service surveys and estimates (USDA Forest Service 1920, 1933, 1948, 1958, 1965, 1973, 1982, 1989; Wadell et al. 1989). Historical wood harvest, from 1910 through 1986, was tracked from primary products, to end uses, to dumps or landfills (Nicholson 1995). Projections of wood harvest and primary product production were made using the models that were used for the 1993 Resource Planning Act (RPA) Assessment Update (Haynes et al. 1995; Ince 1994). These projections were made by the North American Pulp and Paper (NAPAP) model and Timber Assessment Market (TAMM)/ATLAS forest sector models. Historical information and projections from NAPAP and TAMM/ATLAS

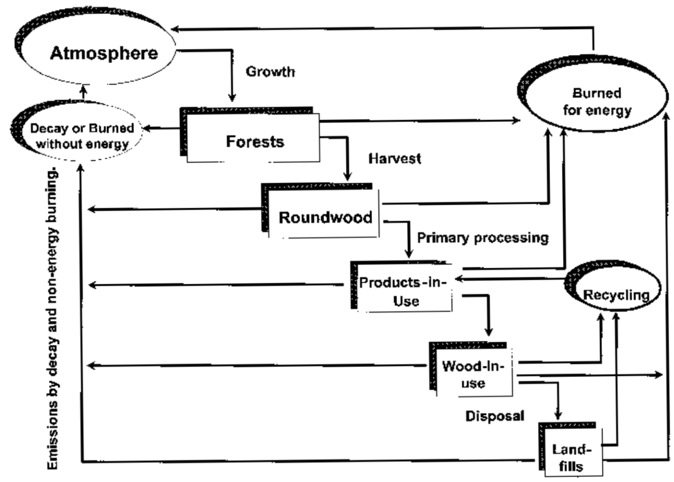

Figure 5.1—Cycling of carbon through wood and paper products.

were processed by the WOODCARB model to make carbon estimates through 2040 for:

- net carbon sequestered in products in use each year (carbon in minus carbon out);
- net carbon sequestered in landfills or dumps each year (carbon in minus carbon out);
- carbon released by burning where useable energy was produced each year; and
- carbon released by decay or burning without energy produced each year.

The NAPAP model simulates operation of markets and projects consumption of pulpwood; use, and change of processing technology; and consumption of pulp and paper. It projects consumption of hardwood and softwood pulpwood, four categories of recycled paper, and production and trade of 13 categories of pulp and paper. The TAMM model and the ATLAS timber inventory projection model simulate the operation of solid wood markets and project consumption of timber, production of lumber and panel products, and end use of lumber and panels in construction, manufacturing, shipping, and other applications (see Mills et al. this volume). The TAMM model also tracks imports and exports of logs, lumber, and panels. The ATLAS model uses NAPAP and TAMM calculations of timber removals to project U.S. forest inventory. The WOODCARB model is an addition to the TAMM model that tracks carbon in all timber removed from U.S. land plus carbon in net imports of logs and wood and paper products.

The following sections explain the methods used to track the flow of carbon in wood from forests, through products and end uses, to landfills and emission by decay or burning.

Carbon Transfer

From Forests to Harvested Roundwood

The carbon in wood harvested each year was estimated through 2040, beginning with wood harvested in 1910 and following each year's wood harvest through to its final disposition. Carbon in wood residue left on harvest sites is not included. Cubic feet of roundwood removed in each of nine U.S. regions is converted to weight of carbon using factors shown in table 5.1 (Birdsey 1992). Carbon in logs imported is added to the roundwood sources, and carbon in logs exported is deducted. The distribution of uses of imported logs is assumed to be the same as the distribution of uses for domestic sawlogs.

Table 5.1—Carbon per unit of roundwood, by region in kg/m^3 (lb/ft^3).

Region	Softwood factors	Hardwood factors
Pacific Northwest-west	242.0 (15.11)	188.4 (11.76)
Pacific Northwest-east	212.9 (13.29)	188.4 (11.76)
Pacific Southwest	242.0 (15.11)	188.4 (11.76)
Northern Rocky Mountains	215.0 (13.42)	191.7 (11.97)
Southern Rocky Mountains	212.9 (13.29)	188.4 (11.76)
North Central	201.0 (12.55)	277.6 (17.33)
North East	194.6 (12.15)	307.7 (19.21)
South Central	270.7 (16.90)	317.5 (19.82)
South East	270.7 (16.90)	317.5 (19.82)

From Roundwood to Primary Products and Residue

Annual historical estimates and projections of detailed product production from the NAPAP and TAMM models were used to divide roundwood consumed into primary product, wood mill residue, and pulp mill residue categories (table 5.2). In most areas, solid wood residues are used almost entirely as raw materials for other processes or are burned for energy. Only a small portion of residues is left to decay or is burned without energy (Powell et al. 1993). Carbon in imports of primary solid wood and paper products is added to each product category, and carbon in exports is deducted.

From Primary Products to End-Use Products and Disposal

Carbon in solid wood products is estimated for nine end-use categories to estimate the time carbon remains sequestered in those products (table 5.3). The TAMM projections are used to divide products into these categories. Pulp and paper products are not tracked to end uses, but the time in use is estimated directly for various primary products. When products are placed in end uses, such as house construction for solid wood and magazine production for paper, some wood or paper is discarded. We assume 8% loss for solid wood products and 5% for paper and paperboard products as they are placed into end uses such as construction or publications. Lost or discarded wood or paper is tracked to recycling, disposal in landfills or dumps, or emission by burning. We estimate ~24 percent of paper and paperboard waste (after recycling) was burned in 1993; this percentage increases to 26 percent for the year 2000 and thereafter (US EPA 1994).

We adapted an equation used by Row and Phelps to estimate the fraction of carbon remaining in end use for each year after the product was placed in use (Row and Phelps, 1996 p. 37). The key parameter in the equation is the half-life for carbon in each end use (table 5.4). The half-life is the time after which half the carbon placed in use is no longer in use. Disposition of carbon after use includes recycling, disposal in landfill or dump, or emission to the atmosphere by burning (with or without energy produced).

The rate of retirement of wood from end uses is constant for a period, then accelerates for a while near the median life, and finally slows down after the median life. Some wood or paper items are expected to have very long lives in uses such as historical buildings, books in libraries, and antiques. The rate of retirement of paper products from use is very fast; the half-life is 1 year or less, except for paper in long-lived publications (free sheet paper), which has a half-life of 6 years.

Carbon Disposal in Dumps and Landfills

The length of time wood, as opposed to paper, remains in end uses may have only a minor effect on the net amount of carbon sequestered in products in the long run. If, when taken out of use, products are disposed of in a modern landfill, the literature indicates that they will stay there indefinitely with almost no decay (Micales and Skog 1997). What may be more important for carbon sequestration or emissions is how much wastewood from discarded wood products or demolition is burned (emitting carbon with or without energy) or how much is recycled (reducing harvest from forests).

Wood and paper sent to landfills (or dumps prior to 1986) includes residue from solid wood mills (in very limited amounts), construction and demolition waste, and discarded paper, paperboard, and solid wood products. These same materials are sometimes burned with or without energy. Prior to 1972, most materials were placed in dumps, where a proportion was burned and contents were more exposed to oxygen and decayed more com-

Table 5.2—Categories of historical and projected wood consumption used to construct estimates of wood carbon use, disposal, and decay.

Historical estimates (1910–1986)	Projections (1986–2040)
Solid wood products and wood mill residue	
Lumber	Hardwood and softwood lumber
Structural paneling	Hardwood and softwood plywood
Nonstructural paneling	Hardwood and softwood in reconstituted panels
Railway ties	Hardwood and softwood miscellaneous products
Miscellaneous products	Hardwood and softwood for roundwood for fuelwood
Roundwood for fuelwood	Hardwood and softwood wood mill residue
Wood and bark mill residue	Hardwood and softwood bark mill residue
Paper and paperboard products and pulp mill residue	
Paper with long use life	Newsprint
Paper with short use life	Coated free sheet
Paperboard	Uncoated free sheet
Sludge and pulp liquor	Coated groundwood
	Tissue and sanitary
	Specialty
	Kraft packaging
	Linerboard
	Corrugating medium
	Solid bleached board
	Recycled board
	Construction paper and board
	Dissolving pulp
	Wood and bark waste
	Sludge and pulp liquor

Table 5.3—End-use categories used to estimate time that carbon remains sequestered.

Solid wood products	Paper and paperboard
Multifamily housing	Use and disposal categories
Mobile homes	Newsprint
Residential upkeep and repair	Boxes
Nonresidential construction	Office paper
Manufacturing	Coated paper
Shipping	Recycled paper categories
Furniture	Old newspaper
Railroad ties	Old corrugated containers
Miscellaneous uses	Mixed paper
Construction waste	Pulp substitutes and high grade deinking
Demolition waste	

Table 5.4—Assumed duration of carbon sequestration in end uses of wood and paper.

End use	Half-life of carbon (years)
Single-family homes (pre-1980)	80
Single-family homes (post-1980)	100
Multifamily homes	70
Mobile homes	20
Nonresidential construction	67
Pallets	6
Manufacturing	12
Furniture	30
Railroad ties	30
Paper (free sheet)	6
Paper (all other)	1

pletely. Legislation then required that dumps be phased out by 1986. Since then, materials have been placed in landfills. Materials in landfills are periodically covered, which prevents oxygen from entering. For dumps, we estimate that 65 percent of waste was burned. We assume the remaining waste decayed evenly during a 96-year period, with a greater proportion of carbon being released as CO_2 than as CH_4 because of a greater mix of oxygen with the materials.

The pattern of landfill decay is markedly different for wood than for paper. A relatively short time after material is placed in a landfill, the material is covered and oxygen is prevented from entering the landfill. While oxygen is available, white-rot fungus can decay lignin to a limited extent. However, the oxygen is consumed rapidly. After the oxygen is gone, only anaerobic bacteria remain. These organisms cannot break down lignin, but they can break down exposed cellulose and hemicellulose. However, anaerobic bacteria cannot reach cellulose or hemicellulose that is enclosed in lignin (Ham et al. 1993; Wang et al. 1994). This means that very little decay of solid wood occurs. Newsprint, which has a lignin content of 20 to 27 percent, is also very resistant to decay. Other papers with less lignin are somewhat more subject to decay. In general, much less than half of the carbon in wood or paper is ever converted to CO_2 or CH_4 (table 5.5) (Micales and Skog 1997).

Not only is the decay of wood and paper highly limited in landfills, but the proportion of carbon emitted as CO_2 is limited to ~40 percent, versus ~60 percent as CH_4, due to the limitation of oxygen and the greater production of CH_4 by anaerobic bacteria. Half of the total CO_2 is emitted in ~3 years, while half the total CH_4 is emitted in ~20 years (Micales and Skog 1997).

The shift to greater CH_4 production in landfills compared with that in dumps is important because CH_4 is 25 times more effective than CO_2 as a heat-trapping greenhouse gas. In our tracking of CH_4 production, we assume 10% of the CH_4 is converted to CO_2 by micro-organisms as it moves out of the landfill. We assume that the proportion of landfill CH_4 that is burned will increase from the current 15% level to 58% by 2040.

Calculating Net Removal of Carbon from the Atmosphere to the United States

One objective of this study is to estimate the combined effect of the forestry sector on net removal of carbon from the atmosphere through the year 2040. This includes sequestration to forests, products, and landfills, and emissions by burning and decay including emissions from imported products. This section will show why net annual sequestration of carbon in U.S. stocks (forests, products, landfills) is greater than the net removal to the United States from the atmosphere by the amount of net imports.

Gross sequestration of carbon to forest trees and soil per year (G) may be expressed as the change in carbon inventory in forests during a year plus carbon in material harvested for products:

$$G = \mathrm{CIC} + \mathrm{HP} \qquad [1]$$

Table 5.5—Estimated maximum proportions of wood and paper that are converted to CO_2 or CH_4 in landfills.

Product type	Maximum carbon converted (%)
Solid wood	3
Newsprint	16
Coated paper	18
Boxboard	32
Office paper	38

where CIC is net sequestration to the inventory of carbon in the forest per year (carbon inventory change). It accounts for any emissions from decay of dead trees or organic material in the soil. It also accounts for emissions from decay or burning of logging residue left after harvesting. HP is harvest and removal of wood carbon for products and wood burning per year. We only include burning of wood after it has been harvested and removed from the forest. Emissions from forest fires are included in the estimate of net sequestration in forests (CIC). Harvesting for products could reduce emissions from fire and increase sequestration in products. This important effect should be the subject of further research.

We now focus on the stock of carbon in the atmosphere, and estimate how the forest sector adds to or decreases the size of this stock. We include the emissions from imports in our variables for emissions from the United States. The rate of removal from the atmosphere per year may be expressed as follows (positive terms represent removal from the atmosphere, negative terms represent additions to the atmosphere):

$$S = G - \mathrm{WB} - \mathrm{ECO_2} - \mathrm{ECH_4} \qquad [2]$$

where S is net removal of carbon from the atmosphere; G is gross sequestration of carbon in forest trees and soil per year, including all growth, even that which is later harvested during the year for products and fuel; WB is emissions of carbon as CO_2 from burning wood, paper, or CO_2 from burning CH_4 for energy production; ECO_2 is emissions of carbon as CO_2 from decay or burning without energy; and ECH_4 is emissions of carbon as CH_4 from decay in landfills, not including CH_4 emitted from wood products in other places such as sewage systems.

The following steps convert equation [2], which expresses annual net carbon removal from the atmosphere to the United States in terms of forest sequestration and emissions, into an equation that expresses the same removal using variables for the annual change in stock of carbon in products in use (P) and stock of product carbon in landfills (L). Let WB = WBWOOD + $WBCH_4$ where WBWOOD is carbon released from burning wood and paper, and $WBCH_4$ is carbon released from burning CH_4 released from landfills.

We may express the net sequestration to the stock of products in use (P) as the amount harvested minus the removal from products in use plus net imports:

$$P = \mathrm{HP} - \mathrm{SL} - \mathrm{WBWOOD} + (I - E) \quad [3]$$

where SL is the amount of carbon shifted to landfills from the stock of products in use each year.

The net sequestration of carbon to landfills each year is the amount shifted from products in use (SL) minus releases:

$$L = \mathrm{SL} - (\mathrm{WBCH_4} + \mathrm{ECO_2} + \mathrm{ECH_4}) \quad [4]$$

By solving equation [4] for SL, substituting in equation [3], and solving for HP, we have

$$\mathrm{HP} = P + L + \mathrm{WB} + \mathrm{ECO_2} + \mathrm{ECH_4} - (I - E) \quad [5]$$

By substituting equations [1] and [5] in equation [2], we obtain an expression for total net sequestration per year that includes the effect of forest growth (CIC), net sequestration to products in use and landfills (P and L), and emissions from burning and landfill decay (WB, ECO_2, and ECH_4):

$$S = (\mathrm{CIC} + P - (I–E) + L + \mathrm{WB} + \mathrm{ECO_2} + \mathrm{ECH_4}) - \mathrm{WB} - \mathrm{ECO_2} - \mathrm{ECH_4} \quad [6]$$

If we focus on the amounts of carbon flows (rather than the different effects of CO_2 and CH_4 in the atmosphere), we may simplify the calculation of carbon removal from the atmosphere.

$$S = \mathrm{CIC} + P + L - (I - E) \quad [7]$$

Equation [7] indicates that net removal from the atmosphere is the sum of net sequestration to carbon in forests, net sequestration to products in use, and net sequestrations to landfills minus net imports.

Annual change in carbon in stocks in the United States may be expressed as

$$\text{Change in stocks} = \mathrm{CIC} + \mathrm{P} + \mathrm{L} \quad [8]$$

To interpret the difference between equations [7] and [8], recall from equation [3] that products in use (P) is harvest (HP) increased by net imports minus emissions and shifts to landfills. So the annual change in stocks includes net imports while annual removal from the atmosphere does not.

Equation [7] does not include carbon emissions from fossil fuels burned for energy in forest sector activities. The sequestration calculated here is the dividend obtained by the forestry activities of the sector. If one were to compare carbon sequestration effects between a forest and a nonforest industry that both provided, say, housing components, one would need to account for not only the fossil fuel emissions of these industries but also any carbon sequestration. The net sequestration effect of using wood housing components is bolstered by the forest regrowth and product or landfill sequestration effects calculated here.

Some may ask why wood burning does not seem to add to sequestration since it replaces fossil fuels and trees grow to absorb the carbon emitted by wood burning. The answer lies in the fact that equation [7] only indicates the net sequestration in one year and does not account for how the value for carbon inventory change (CIC) may be higher in a future year or years as a result of harvesting and burning wood in the current year. A forest growth and yield model is needed to evaluate the degree to which the CIC value is higher in the future due to harvest and use of wood for energy in the current year. In the analysis for this study, we used the ATLAS inventory growth and yield model to calculate actual future increases in forest growth.

Calculating the Greenhouse Gas Effect of Net Carbon Removal to the United States

The greenhouse gas effect of net carbon sequestration by the forest sector is determined in part by whether carbon is emitted to the atmosphere as CO_2 or as CH_4. A CH_4 molecule is 25 times more effective in trapping heat than a CO_2 molecule (U.S. Congress OTA 1991). However, CH_4 lasts an average of 10 years in the atmosphere, while CO_2 lasts at least 50 years before breaking down. The long-term greenhouse effect of a CH_4 molecule has been estimated to be ~21 times greater than the effect of a CO_2 molecule (U.S. Congress OTA 1991). To approximate the greenhouse gas effect of net carbon removal (S), we need to convert carbon emitted as CH_4 (ECH_4) to its weight in terms of the heat trapping effect of carbon in CO_2. That is, an atom of carbon in CH_4 results in 21 times more heat trapped than an atom of carbon in CO_2.

$$S_g = (\mathrm{CIC} + P - (I - E) + L + \mathrm{WB} + \mathrm{ECO_2} + \mathrm{ECH_4}) - \mathrm{WB} - \mathrm{ECO_2} - 21(\mathrm{ECH_4}) \quad [9]$$

$$S_g = \mathrm{CIC} + P - (I - E) + L - 20(\mathrm{ECH_4}) \quad [10]$$

where S_g is net carbon removal after converting the CH_4 emissions term to CO_2 equivalent weight.

About 40% of the carbon from wood and paper decaying in landfills is emitted as CO_2 and about 60% as CH_4. The CO_2 is released quickly, while oxygen is present, and the CH_4 is released very slowly after oxygen is depleted (Micales and Skog 1997). Since half the carbon is emitted

Table 5.6—Estimates of harvested wood carbon sequestered, emitted, and consumed in U.S. annually in Tg (historical reconstruction 1910 to 1980, with projections to 2040 [RPA Base case]).

Year	Added to products in use	Added to landfills	Emitted by burning with energy	Emitted by decay or burning without energy	Total consumed each year
Historical reconstruction					
1910	24.3	1.1	88.4	10.6	124.4
1920	22.9	3.1	51.9	14.7	92.6
1930	12.8	4.1	44.6	15.5	77.0
1940	14.0	5.3	35.0	20.4	74.7
1950	13.6	6.3	37.4	25.5	82.8
1960	9.0	7.1	34.6	30.6	81.3
1970	12.4	9.2	32.8	35.9	90.3
1980	11.8	27.9	48.1	19.2	107.0
Base Case projections					
1990	26.0	33.4	74.4	11.4	145.2
2000	25.0	32.5	88.1	14.3	159.9
2010	24.6	38.0	96.8	15.3	174.7
2020	25.6	42.6	103.0	16.4	187.6
2030	24.4	47.0	109.5	17.1	197.9
2040	22.9	50.8	119.0	17.5	210.2

as CH_4, converting it to CO_2 could have a notable effect in raising the carbon sequestration by the forestry sector.

Results

Several key factors determine the pattern of historical and projected carbon sequestration and emissions from wood and paper products.

The total carbon contained in roundwood harvest plus net imports declined between 1910 and 1940 (from 124 to 74 Tg/year) in part as a result of steadily decreasing fuelwood use. After the 1940s the amount of carbon in roundwood doubled by 1995: 74 Tg/year to 150 Tg/year. Total carbon in roundwood and net imports is projected to increase to 210 Tg/year by 2040 as indicated using the 1993 RPA Base case projections for the U.S. forest sector (table 5.6; fig. 5.2).

Since the early 1900s the use of roundwood in primary products (lumber, panels, paper and paperboard, fuel) has shifted from solid wood products and fuelwood, to a mix of products that includes an increasing proportion of paper products and more burning of residue from solid wood products mills and black liquor from pulp mills.

Even though carbon held in solid wood products is projected to double between 1950 and 2040 (30 to 60 Tg), carbon in pulpwood used in paper production will

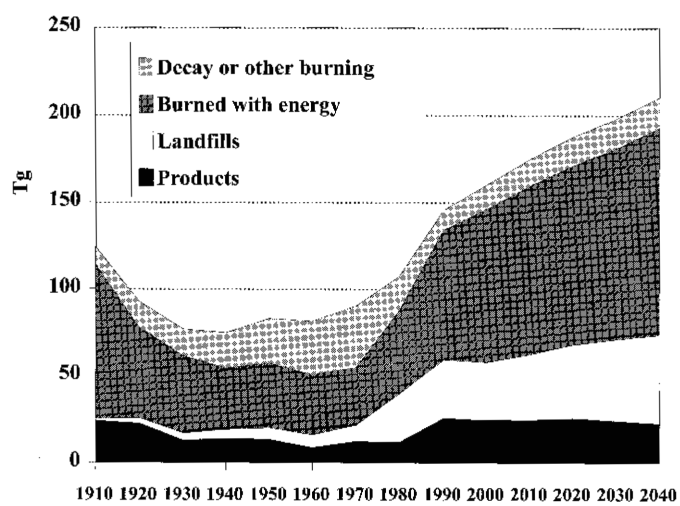

Figure 5.2—Annual net sequestration of harvested wood carbon in products and landfills, and annual emissions from wood burning with energy, and wood and paper decay and other burning in the United States, 1910 to 1993, with projections to 2040.

increase 600 percent (to 81 Tg) by 2040. Burning of wood residue and black liquor has also increased relative to solid wood uses, from 1 Tg in 1910 to 21 Tg in 1990 and will be 31 Tg in 2040. Fuelwood use, reaching a low of 3 Tg in 1970, is projected to surpass its 1920 level by 2040 and remain slightly higher than burning of wood residue and black liquor (fig. 5.3).

Table 5.7—United States net carbon accumulation, emission, net imports, and removal from the atmosphere by year[a].

	Net carbon flux (Tg)					
	1990	2000	2010	2020	2030	2040
Change in forests, CIC	274	189	192	176	166	161
Change in products in use, *P*	26.02	24.99	24.51	25.58	24.27	22.86
Change in landfills, *L*	33.38	32.48	39.37	42.53	46.89	50.74
Wood burning, WB	74.38	88.07	96.58	102.83	109.27	118.86
Emitted CO_2, ECO_2	11.43	14.02	14.83	15.77	16.49	16.98
Emitted CH_4 from landfills, ECH_4	0	0.23	0.5	0.61	0.62	0.55
Change in stock of carbon[b]	333.4	246.47	255.88	244.11	237.16	234.6
Net imports of wood products, paper, and paperboard (*I* - *E*)	2.33	3.26	3.67	3.87	2.84	1.50
Removal from atmosphere, *S*[c]	331.07	243.21	252.21	240.24	234.32	233.1
Removal from atmosphere in CO_2 equivalents, S_g[d]	331.07	238.61	242.21	228.04	221.92	222.1

[a] *Base case projections*
[b] Change in stock of carbon = CIC + *P* + *L*
[c] S = CIC + P - (I - E) + L (net carbon removal from atmosphere)
[d] S_g = CIC + P - (I - E) + L – 20(ECH_4) (net carbon removal from atmosphere in CO_2 equivalents)

Overall, the rate of net sequestration of carbon to products in use and landfills increased ~170 percent between 1970 and 1990—from 22 to 59 Tg/year. This net sequestration reflects disposal and decay of products taken out of use, sent to landfills or burned; and decay of wood and paper in landfills. This increase was due in part to the increase in product consumption; roundwood use increased 51 percent between 1970 and 1991, 35 to 53 × 10^7 m^3 (12.5 to 18.7 × 10^9 ft^3) (Heath and Birdsey 1993). It is also due to a sharp increase in the rate of accumulation of carbon in landfills with the shift from dumps to landfills in the 1970s and 1980s. Net accumulation in dumps or landfills increased from 9.2 Tg/year in 1970 to 33.4 Tg/year in 1990. This increase in net accumulation was due to virtual elimination of open air burning in dumps and a decrease in the rate of decay of wood and paper in landfills compared with that in dumps.

Using the 1993 RPA Base case projections for the forest products sector, the annual rate of carbon sequestration to forest trees, understory, floor, and soil is projected to decline from 274 Tg in 1990 to 161 Tg in 2040 (table 5.7) (Birdsey and Heath 1995). This trend reflects a slowdown in the rate of accumulation in the North as forests reach an age of slower tree growth and slower increases in soil carbon, and a reduced harvest on public land in the West along with more intensively managed areas of former old growth. It also reflects increased management intensity in the South, where accumulation is balanced by removals (Birdsey and Heath 1995).

The annual rate of carbon accumulation in landfills or dumps and products is projected to increase from 59 Tg in 1990 to 75 Tg in 2040. This is due entirely to the increasing

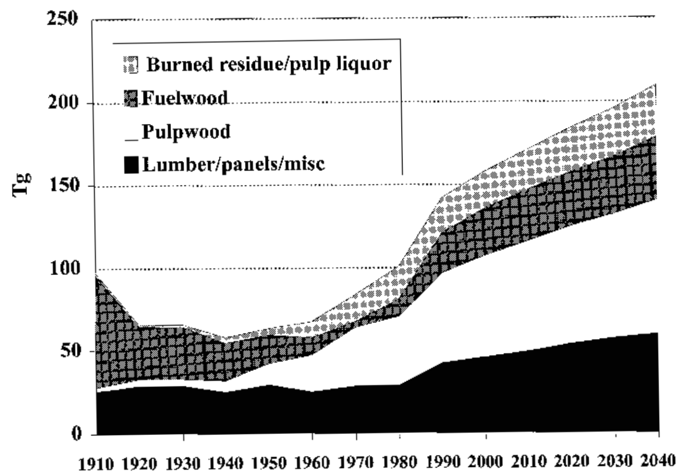

Figure 5.3—Initial product uses of roundwood harvested in the United States, 1910 to 1993, with projections to 2040.

rate of accumulation in landfills. The net annual sequestration to products in use actually decreases slightly from 26 Tg in 1990 to 23 Tg in 2040. This decline is due in part to the increasing proportion of wood that is used in paper products, which have a shorter use-life than do solid wood products.

Our estimate of 59 Tg carbon added to landfills, dumps, and products in 1990 is close to the estimate of Winjum et al. (1998) of 57 Tg stored in commodities for five years or more for the United States.

Carbon emissions from burning with energy production are projected to increase as a result of notable increases in burning of black liquor and roundwood (directly from forests) for fuel. Black liquor and roundwood carbon emissions increase from 54 to 92 Tg between 1990 and 2040. Burning of mill residue and other wood or paper waste increases emissions from 20 to 27 Tg during the same period. Emissions from burning without energy production and from decay are projected to increase from 11 Tg to 18 Tg between 1990 and 2040. In total carbon, emissions increase from 86 Tg to 137 Tg between 1990 and 2040.

Our estimate of 86 Tg of total emissions in 1990 is identical to the estimate of Winjum et al. (1998) for emissions from all sources in 1990.

In 1990, we were adding carbon to the wood and paper product stocks at the rate of 59 Tg per year. This rate is projected to increase to 74 Tg per year by 2040 (tables 5.6 and 5.7). If we add sequestration to forest trees, understory, floor, and soils, the rate of sequestration to U.S. carbon stocks is 333 Tg/yr in 1990 and 235 Tg/year by 2040.

The annual net removal of carbon from the atmosphere to the United States is slightly less than the accumulation in stocks due to net imports supplementing U.S. stocks. Net removal of carbon from the atmosphere is 331 Tg for 1990 and is projected to decline to 233 Tg by 2040. Net removal measured in CO_2 equivalent effect on the atmosphere is 331 Tg for 1990 and is projected to decline to 222 Tg by 2040. In 1990, the total carbon removal from the atmosphere to U.S. forests and forest products was 24 percent of the U.S. fossil fuel carbon emissions level of 1,367 Tg (331/1367).

Our estimates of the cumulative fate of carbon in the United States since 1910 (including net imports) are shown in figure 5.5. We estimate total carbon in wood and bark used for products and fuel between 1910 and 1990 at 7.8 Pg (where Pg is 1 billion metric tons). We estimate 2.1 Pg accumulated in products and landfills, 4.0 Pg in wood and bark burned for energy, and 1.7 Pg in emissions. Total accumulation over the projection period, from 1990 to 2040, is 9.0 Pg. Accumulation in products and landfills is projected to be 3.2 Pg between 1990 and 2040 for a total of 5.3 Pg over the period 1910 to 2040.

Conclusions

Our projections indicate we have accumulated 2.1 Pg of carbon in the stock of wood and paper products in use and in landfills and dumps in the United States between 1910 and 1990. This is substantial compared with the 1992 stock of carbon in forest trees (13.8 Pg) and in forest soil (24.3 Pg).

Annual sequestration to product carbon stocks in the United States are slightly greater than annual removal of carbon from the atmosphere. Forest, product, and landfill stocks increased 333 Tg/yr in 1990 while net sequestration to forests, products, and landfills was 331 Tg/yr. This difference is because net sequestration to stocks includes net imports while annual removal from the atmosphere does not. Net removal from the atmosphere may be increased by burning CH_4 from landfills to convert it to CO_2, which has less greenhouse effect.

The choice of accounting method (measuring stock changes, or measuring removals from the atmosphere) could determine how a country would count carbon imports and exports in offsetting greenhouse gas emissions for the purpose of meeting goals under international agreements. This in turn could influence forest and industry management in the U.S. If credit is given for increasing stocks, a country would seek to boost imports and restrain exports. If credit is given for increasing removals from the atmosphere, there would be an emphasis on increasing carbon sequestration in forests and in products from domestic forests that may be aided by increasing exports and restraining imports. There may also be more emphasis on decreasing methane emissions from forest products decay in landfills.

By recognizing what has caused changes in sequestration to carbon stocks in wood products and forests we can identify some ways sequestration can be increased even more. We can increase sequestration to the stock of carbon in products, landfills, and forests while maintaining the same aggregate consumption of wood and paper products by the following actions:

- shifting product mix to a greater proportion of lignin-containing solid wood, paper, and paperboard products, which decay less in landfills;
- increasing product recycling; and
- increasing product use life.

Carbon dioxide equivalent emissions would also be reduced by the actions noted. Emissions would also be reduced by burning more landfill CH_4 in place of fossil fuels.

It may be possible to increase sequestration while increasing product consumption above projected levels but this would be determined by the certain effects of such an increase not assessed in this study:

- How much would annual carbon inventory change in the forest increase in the future as a result of increased harvest today?
- How much would manufacturing emissions change due to substitution of wood and paper for nonwood products?
- How much would emissions from forest fires decrease due to reduction in fuels available for fires?

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Soil Carbon Accounting and Assumptions for Forestry and Forest-Related Land Use Change

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Introduction

Comprehensive, large-scale carbon accounting systems are needed as nations agree to work toward reducing their greenhouse gas (GHG) emissions. However, adopting a standard accounting system is difficult because multiple science and policy uses for such a system help fuel the debate about the nature of an appropriate system. Accounting systems must address all major sources and sinks of GHGs, or more pragmatically, focus on subsets of important sources and sinks and feature transparent, fundamental rules that may be adopted easily by all nations. Here, we review some issues in carbon accounting of a major GHG sink: forest soils, at a national scale. Specifically, we concentrate on how land use change and harvesting affect forest soil carbon, and how those effects may be described clearly in an accounting system that is easy to use.

Organic carbon in soil below the forest floor is one component of forest carbon that is particularly contentious. Measuring soil carbon is time-consuming, costly, and operationally difficult, partly because variability in soils tends to be high, requiring many samples to statistically test results. Relationships between easily characterized aboveground vegetation and belowground soil carbon may be weak because soil carbon may have been affected by past land use, long after visual traces of the previous use disappear. However, carbon pool size alone makes forest soils quite important, despite the uncertainties (EIA 1997; US EPA 1998). Soils of the world are estimated to contain twice the amount of carbon as in the atmosphere or vegetation (Bouwman 1990).

The accounting frameworks described in the global guidelines for national greenhouse gas inventories released by the Intergovernmental Panel on Climate Change (IPCC/OECD/IEA 1995, revised in 1997) increasingly discuss soil carbon, thereby reflecting the importance of accounting for carbon in soil. In the United States, Birdsey and Heath (1995) presented forest carbon estimates, including soil carbon, in a technical document (Joyce 1995) accompanying a larger assessment framework: the USDA Forest Service analysis for the Forest and Rangeland Renewable Resources Planning Act (RPA) Assessment (USDA FS 1994). These accounting systems feature two main components: input measures or samples to characterize forests, and a core of assumed relationships to estimate the amount of carbon in that forest. Carbon is generally described as a function of forest age, area, volume, or biomass.

In this chapter, we discuss the accounting system by Birdsey and Heath (1995) used by the RPA and the accounting system of the IPCC National Greenhouse Gas Inventories (1997) for soil carbon in the forest sector and land use changes involving forests. Basic assumptions are compared in light of new scientific studies on forest soil carbon. We outline important components in the accounting frameworks with an emphasis on land use change activities such as afforestation, deforestation, and reforestation. We then review recent scientific developments that affect soil carbon assumptions used to calculate carbon estimates.

Forest Carbon Accounting

We use the phrase “soil carbon” to mean soil organic carbon in horizons beneath (and not including) the forest floor. Usually soil beneath the forest floor is called mineral soil. Some soils developing under a waterlogged condition may contain a high level of organic matter, and these soils are called organic soils, or Histosols. Thus we can differentiate between soil organic carbon in mineral soils and soil organic carbon in organic soils. Some soils also contain a great deal of carbon in inorganic form such as carbonates. However, inorganic carbon is relatively inert and therefore we do not include it in this study.

Carbon accounting quickly becomes complicated in practice, because we are most interested in carbon flux, which may be calculated as change in successive carbon stocks (inventories). At the simplest level, two variables are needed to calculate net carbon flux in forests: area and total carbon per area. Multiplying area and carbon per area yields total carbon inventory stored in forests. Net carbon flux is then estimated by the difference in total carbon estimated at two consecutive times divided by the length of time between inventories. However, there are a number of methods to model carbon inventory and flux. The methods vary by data requirements, system definition, boundary conditions, and even identity of carbon pools. The two methods we discuss focus principally on estimating stocks of carbon, and flux is simply the annual difference in stocks.

Data and information issues make forest carbon accounting particularly difficult. Ideally, a comprehensive accounting system would provide the best estimates for GHG emissions associated with forests. A comprehensive system would include these components at the beginning of each year: living and dead tree biomass carbon, carbon in seedlings, understory, forest floor, root, and soil carbon. No nation measures all these components, although some nations do measure many items strongly related to these components. Further, scientists do not agree on generalized assumptions that may be used as a way to convert measured data (such as area and volumes) to carbon. Without scientific consensus on assumptions, some nations may prefer to exclude incomplete information, while other nations may have adequate information. No matter how detailed the information, the goal is to develop estimates of the area of forest and carbon per unit area.

The comparison of the two accounting methods does not include a quantitative estimate formed using the IPCC method. A summary of U.S. carbon totals from previous estimates is included here to provide some perspective on the magnitudes involved. Forests in the conterminous United States were estimated to contain about 37.7 billion metric tons of carbon in 1992, sequestering 127 million metric tons per year in soils and forest floor, and 84 million metric tons per year in live vegetation (see table 4.2, Birdsey and Heath 1995). These results included estimates of forests of very low productivity, which tend to be located in arid or mountainous regions, and are for the most part not managed commercially for timber. Productive forestland available for harvest is called timberland. Carbon estimates in 1992 for timberland, including timberland in Alaska, are 34.3 billion metric tons, sequestering 84 million metric tons per year in soils and forest floor, and 74 million metric tons per year in live vegetation (see table 4.3, Birdsey and Heath 1995). The two tables are not strictly comparable because of definitional changes over time. For instance, the carbon estimates on timberland during the period 1977–1992 are noticeably affected by Congressional designation of some timberland as Wilderness—an example of how land use change can affect apparent carbon budgets.

Soil Carbon Accounting Systems For Forest and Land Use Change

A comparison of two carbon accounting methods designed for national-level totals can usefully illustrate some of the links among the state of scientific understanding, model assumptions, and assessment priorities. The first method was developed to estimate total carbon inventory of U.S. forests, with emphasis on change in storage brought about by growth and management activities. The second method is from an international effort designed to be generally applicable. It is focused on determining forest GHG emissions induced by human activities.

The two methods are not simply alternate approaches to estimating the same values, but there are some parallels in assumptions and goals. Comparisons are essentially qualitative overviews of the conceptual organization of the two methods. We emphasize assumptions and approaches to modeling land use change, especially afforestation, deforestation, and reforestation. Although the focus of our study is on soil organic carbon, we also discuss carbon in other components of forests because often the soil carbon information inextricably depends on other forest components.

Carbon Estimates Used by the RPA Assessment, 1995

The carbon accounting method of Birdsey and Heath (1995) was used for the 1995 RPA assessment and included comprehensive estimates for carbon in all components of U.S. forests. These were developed specifically for U.S. forestland and designed to utilize the extensive base of forest information in this country. We first discuss some basic assumptions of the method, and we then discuss accounting for effects of changes in land use.

The estimates were based on forest inventories conducted by the USDA Forest Service Forest Inventory and Analysis Program (Hansen et al. 1992; Woudenberg and Farrenkopf 1995). The inventory survey and associated sampling errors are designed for measuring total timber volume over an aggregated forest area. Forest areas are estimated because they are needed to calculate volumes, but the design of the survey does not require designating sampling errors for area or soil carbon. As a result, forest inventory data provide good above-the-stump information, yet are also useful for deriving belowground information calculated as assumed functions of collected data. The inventory does not directly measure soil carbon.

The carbon model is based on aggregations of forests within each of nine regions of the United States. Each aggregation is called a management unit. Forest type, ownership, and sometimes productivity and previous land use delineate each management unit. The forest inventory for each management unit includes number of hectares and average volume by age class. Soil carbon per hectare is estimated for management units according to empirical relationships specific to management unit characteristics (Birdsey 1992; Plantinga and Birdsey 1993). Similar estimation procedures are also established for other forest carbon pools, including carbon in trees. Net annual soil carbon flux is calculated by multiplying hectares of forest by carbon per hectare in each of two consecutive inventories and dividing the difference by length of the period.

The number of hectares of forest inventory may change as land use changes between forest and non-forest vegetation. Such area change influences on flux are allowed in this method to provide an accurate accounting of carbon in the forest sector. If land was transferred into the agricultural sector from the forest sector, then it is assumed the additional hectares are accounted for in agriculture. This apparent loss of forest soil carbon to the atmosphere is really only a transfer to a different sector. Similarly, if forested area increased during the interval, this accounting method produces an effect of additional carbon sequestration from the atmosphere. In the 1992 and prior inventory estimates, areas are based on historical estimates from forest inventories; in the projected years, areas are based on land use projections (Alig et al. 1990) used in the RPA Timber Assessment (Haynes et al. 1995).

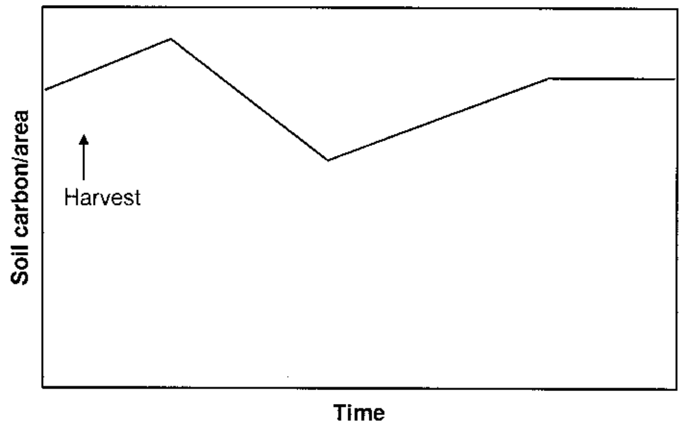

Figure 6.1—Generalized trajectory of forest mineral soil organic carbon following clearcut harvest. (Adapted from Moore et al. 1981.)

RPA Accounting System and Land Use Change

Assumptions about the dynamics of soil carbon over time are discussed in Plantinga and Birdsey (1993) and Birdsey (1992); these include effects of both previous land use and harvesting. Initial soil carbon estimates for forests developing on cropland and pasture–that is, the land use is changing from cropland or pasture to forest–were derived from regression equations for soil organic carbon in Burke et al. (1989). Regional estimates were based on mean annual temperature and mean annual precipitation for each of the regions, assuming percent clay and silt were equal to 20 percent and 40 percent. Birdsey (1992) developed a comparable regression equation for forestland, with the results equal to soil organic carbon of mature forests, which was assumed to occur at age 50 in the South and at age 55 in other regions. With these three base soil carbon estimates (that is, forest originating on cropland, pasture, or forestland) developed by region, the dynamics of soil carbon with forest growth were functions of previous land use and time. Forests regenerated on pasture were assumed to start (at forest stand age 0) with soil carbon characteristic of pasture, and then increase linearly as forests aged to the amount of soil carbon found in mature forest stands of that region. Soil carbon of forests regenerated on cropland was estimated similarly. After clearcut harvest (at forest stand age 0), soil carbon is assumed to equal the calculated base carbon estimate, decline up to 20 percent (Woodwell et al. 1984; Pastor and Post 1986) over a 10–15 year period following harvest, and accumulate gradually to a base forest carbon by maturity (approximately 50 years). These qualitative trends in soil carbon are illustrated in figures 6.1 and 6.2 for a clearcut with reforestation and harvest with a non-forest interval before regeneration, respectively.

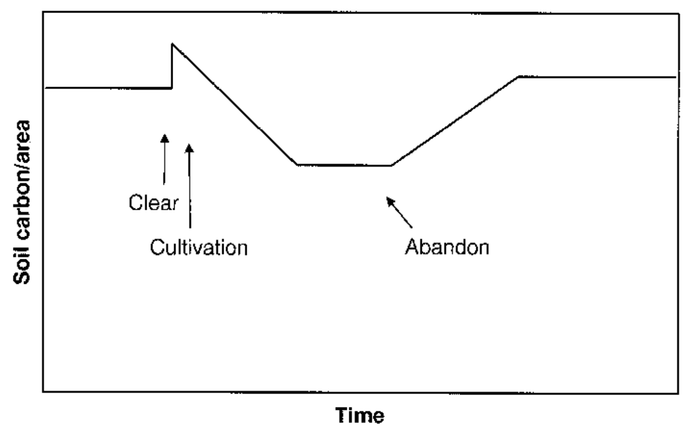

Figure 6.2—Generalized trajectory of forest mineral soil organic carbon after clearing, cultivation, and forest regeneration following abandonment. (Adapted from Moore et al. 1981.)

Management units are not partitioned into areas of previous land use, so we cannot simply adopt soil carbon estimates for forests originating on cropland, pasture, or forestland. We currently have only general estimates of previous land use over an aggregated area. Therefore, weighted averages of soil carbon were calculated, based on percentages of previous land use on which current forest were regenerated. These percentages were estimated using various USDA Forest Service inventory statistics, coupled with assumptions about management, ownership, and regional influences (Birdsey, personal communication). It was the weighted soil carbon equations that were used in the analysis.

The effect of the weighting procedure is illustrated in figure 6.3, which shows the soil carbon trajectories for planted pine on productive sites of different previous

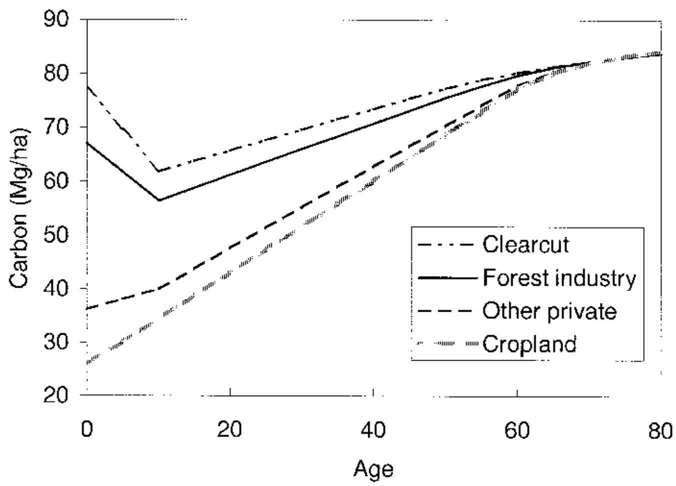

Figure 6.3—Soil organic carbon (0-1 m) by forest stand age for planted pine on high productivity sites in the southeastern United States, based on old assumptions. The estimates are weighted averages based on percentages of forest in previous non-forest land use. (This is but one example set of equations available for each combination of region,forest type, and ownership.) Source: Personal communication. 1994. Data from Richard Birdsey, Program Manager, USDA Forest Service Northeastern Research Station, Radnor, PA 19087.

land use in the southeastern United States. One of the trajectories illustrates how soil carbon per hectare would accrue by age for forestland that had been clearcut and replanted. The second trajectory represents soil carbon per hectare for forest growing on land that was previously cropland. However, neither of these trajectories accurately represents the aggregate soil carbon trajectory for a mixture of previous land uses. On forest industry lands, for example, 80 percent of the forestland was previously forest, while 20 percent was cropland. The soil carbon trajectory is an average of the carbon on clearcut forests and carbon on cropland, weighted by percentage of land in each use. This average trajectory is labeled "Forest industry." The trajectory for "Other private" ownership is based on the estimate that 80 percent was previously in cropland and 20 percent was previously forested. Soil carbon trajectories of other forest types and regions were calculated using the same weighting procedure.

Previous land use heavily influences soil carbon as illustrated in figure 6.3. In this example, soil carbon per hectare at age 0 ranges from 26 Mg C per ha on cropland to 78 Mg C per ha on a clearcut. After about 65 years of forest development, soil carbon is about 80 Mg C per ha regardless of previous land use. Age is often difficult to determine in naturally regenerated forests, and forestland is often not fully occupied by trees. Although this figure illustrates a relationship between soil carbon and forest age, the accounting method used by RPA often employs a relationship between forest merchantable volume and soil carbon. However, relationships with forest age are used for stands less than 15 years of age, when merchantable volumes are zero or close to zero. Volume is thought to more accurately reflect the level of soil carbon when characterizing older stands (Plantinga and Birdsey 1993; Birdsey and Heath 1995).

The IPCC Method of Estimating Carbon, 1997

One of the objectives of the IPCC/OECD/IEA Programme on National Greenhouse Gas Inventories (IPCC/OECD/IEA 1995, 1997) was to develop a default methodology, with the concurrence of the international scientific community, which nations could follow or use as guidelines to report GHG emissions and sinks. A goal was to be both extensive and simple. This would produce a methodology appropriate for use by any nation, yet estimates could be determined even with limited data. Nations are strongly encouraged to use local information if doing so would increase accuracy of estimates. We review the methodology of the Land Use Change and Forestry (LUCF) section of the guidelines (IPCC/OECD/IEA 1997) by first discussing some basic assumptions. We then address some issues of accounting for carbon under changes in land use.[1]

Classification of land and activities on that land are important first steps in the IPCC guidelines. Areas of forests that are currently not significantly disturbed by humans are excluded from calculations. That is, areas of land which feature a carbon flux of approximately zero are ignored for carbon accounting purposes. The distinction between forestry and other agricultural activities also distinguishes how carbon is counted. The LUCF section of the guidelines includes land use change and carbon emissions from agricultural activities. There is a separate extensive section on agriculture; however, it focuses on nitrous oxide emissions from agricultural soils, emissions from agricultural burning including prescribed burning of savannas, and methane and nitrous oxide emissions from domestic livestock. Prescribed burning of savannas is handled in the agriculture section, yet burning of savannas for the purpose of changing land use is handled in the LUCF section.

The IPCC guidelines categorize forestland as tropical, temperate, or boreal. We review the overall methodology,

[1] *At the third Conference of the Parties in Kyoto, Japan, the Parties agreed to count forest carbon from afforestation, deforestation, and reforestation since 1990. However, definitions for these three terms are still under discussion so we review the current published guidelines.*

Table 6.1—Headings of major categories for changes in forest and other woody biomass aboveground carbon stocks suggested by IPCC (1997) for calculating national greenhouse inventories.

Latitude	Woody biomass stocks		Changes in harvesting	Conversion & abandonment
Temperate	Plantations	Douglas fir	Specified by user	Coniferous
		Loblolly pine		Broadleaf
	Commercial	Evergreen		Grasslands
		Deciduous		
	Other			
Boreal	ND	ND	Specified by user	Mixed
				Broadleaf/coniferous
				Coniferous
				Forest-tundra
				Grasslands/tundra
Tropical	Plantations	*Acacia spp.*	Specified by user	Wet/very moist
		Eucalyptus spp.		Moist, short dry season
		Tectona grandis		Moist, long dry season
		Pinus spp.		Dry
		Pinus caribaea		Montane moist
		Mixed hardwoods		Montane dry
		Mixed fast-growing hardwood		Tropical Savanna/grassland
	Other forests	Moist		
		Seasonal		
	Other			

ND = No default specified.
Source: IPCC/OECD/IEA 1997.

but we focus on temperate forests because they constitute the majority of U.S. forestlands. Aboveground and belowground carbon pools are estimated separately. Temporal responses to perturbations differ for two systems: several decades may be needed for soil carbon to respond to change and stabilize, while only a few years may be adequate to describe responses of aboveground biomass to the same changes.

The default approach for estimating aboveground carbon inventories features biomass tabulated by the categories of forest and other woody biomass stocks, forest and grassland conversion, and abandonment of managed lands. Each of these categories is further divided by vegetation types under tropical forest and grasslands, temperate forest and grassland, boreal forest and tundra, and other. The category "forest and other woody biomass stocks" features more specific forest types. The default headings for these three categories are displayed for comparison in table 6.1. The categories, with the exception of harvesting which has no defaults specified, are based on vegetation type.

Soil carbon emissions are tabulated by the categories of soil carbon emissions from mineral soils, organic soils (Histosols), and liming of agricultural soils. Liming is not a common treatment in forestry in the United States, so it is not addressed here. Organic soils are commonly bog soils and are most prevalent in localized areas of the United States, with the largest contiguous areas in Minnesota, Louisiana, and Florida. Most of the change in organic soils is due to cultivation for agriculture, particularly vegetable crops. The default headings for mineral and organic soil carbon categories are displayed in table 6.2. Note that soil carbon is classified by climate, soil type, and then vegetation and management system. Although the forest-related vegetation/management system is broadly categorized (for example, one category is forest), IPCC recommends that forest and grassland management systems be subdivided into relevant categories.

Soil carbon emissions are estimated by first multiplying the current area of a given vegetation/soil type/management system by the amount of soil carbon estimated in each hectare to produce total soil carbon stock. Soil carbon flux at a designated time in the past is calculated for the same land base using the areas at the previous time, and then subtracting the previous soil carbon total from the current soil carbon total. Dividing by the length of the period between measurements converts net soil carbon flux to an average annual basis. The calculation for net soil carbon flux is expressed in equation form as:

Table 6.2—Headings of major categories suggested by IPCC (1997) for calculating changes in soil carbon for areas undergoing land use change·

	Soil Carbon Change In Mineral Soil			Carbon Emissions From Organic Soils	
Latitude	Climate	Soil type	Vegetation and management systems	Climate	Soils use
Temperate	Cold, dry	High clay Low clay Sandy Volcanic Wetland	ND	Cool Warm	Upland crops Pasture/forest Upland crops Pasture/forest
	Cold, moist	High clay Low clay Sandy Volcanic Wetland	Forest Forest set-aside		
	Warm, dry	High clay Low clay Sandy Volcanic Wetland	ND		
	Warm, Moist	High clay Low clay Sandy Volcanic Wetland	Forest Forest set-aside Reverted forest		
Boreal	ND	ND	ND		
Tropical	Dry	High clay Low clay Sandy Volcanic Wetland	Savanna	All	Upland crops Pasture/forest
	Moist	High clay Low clay Sandy Volcanic Wetland	Forest/woodland Plantations		
	Wet	High clay Low clay Sandy Volcanic Wetland	Forest/woodland Agroforestry Plantations		

ND = No default specified.

[1] Source: IPCC/OECD/IEA 1997

[2] Management systems involving forest. Examples of agricultural management systems not listed in this table are small grain with continuous cropping, hay improved pasture, successional grassland, irrigated cropping systems and intensive grain production.

$$\text{Soil carbon flux} = \left[\sum(\text{Area}_{i,t} \times \text{C}_i) - \sum(\text{Area}_{i,t-\text{LP}} \times \text{C}_i)\right] / LP \qquad [1]$$

where $\text{Area}_{i,t}$ = total area of the *i*th vegetation/soil type/management system at time t, C_i = soil carbon per area of the *i*th vegetation/soil type/management system, and LP = length of period in years.

If area of each vegetation/soil type/management system does not change over the period, then soil carbon emissions are zero. Total land area across all vegetation/soil type/management system categories should always remain constant.

Tables 6.1 and 6.2 are presented here as examples of the main variables of interest currently included in the IPCC guidelines. However, they also show the dichotomy induced in the accounting system by the characteristics of the underlying inventory system. Aboveground carbon is based on volumes of wood growing in or harvested from forests, while soil carbon is based on hectares categorized by land use. This system becomes unwieldy when carbon accounting is focused on only a part of the land base. That is, accounting for only afforested or deforested hectares can be difficult when independent variables predicting the aboveground and belowground portions do not coincide.

IPCC Accounting System and Land Use Change

Estimates of mineral soil carbon per unit area are based on default values provided in the IPCC guidelines for native vegetation by climatic region and soil type. These estimates are then multiplied by a use factor, tillage factor, and an input factor. The tillage factor is used only for agricultural soils. When temperate native soils are cultivated, soils are assumed to lose 30 percent of the soil carbon in the 0–30 cm soil layer, with the exception of wet soils that are assumed to lose 40 percent. Forested lands cleared and put under long-term cultivation are assumed to lose 30 percent of the soil carbon (Davidson and Ackerman 1993). Soils under long-term cultivation, but then set aside and not managed for less than 20 years, are assumed to contain 20 percent less soil carbon than native soils; soils set aside and not managed for more than 20 years are assumed to contain 10 percent less soil carbon than native soils. However, set-aside land apparently does not include land planted to forests. The current default accounting does not include accumulation of carbon in soil in plantations established on previously unforested (for at least the last 50 years) lands. Default values are less likely to be used by countries with significant activities that can affect soil carbon such as establishing plantations, for example (see note on page 5.15 of Volume 3, IPCC/OECD/IEA 1997). Soils under improved pasture gain 10 percent more carbon than the same soil under native vegetation, an assumption attributed to work by Fisher et al. (1994); Cerri et al. (1991); and Grace et al. (1994).

Organic soils are handled differently than mineral soils. Estimates of losses of organic soils under introduced pasture and forests average 25 percent of the loss rate under crops. This translates into an annual loss rate of 0.25, 2.5, and 5.0 Mg C per ha per year, for pasture or forestland intensively managed in cool temperate, warm temperate, and tropical areas. This simple relationship is a good illustration of the stated IPCC goal of generally applicable methods. Emissions are calculated for only those hectares currently under intensive use by multiplying number of hectares in each land use by the default annual loss. Organic soils under native vegetation are not included because they are assumed to have stable or increasing carbon stocks.

The current IPCC method employs simple assumptions about the dynamics of soil carbon. Soil carbon is presumed to tend toward equilibrium after many years under a specific land use. Spatial and temporal bounds are set as the top 30 cm and within 20 years. Only soil carbon in the 0–30 cm soil depth is considered for both mineral and organic soils. This area typically has the greatest concentration of carbon and the fastest response time to disturbance. The default guidelines suggest that soil carbon stock estimates include carbon in the forest floor (litter layer), as well as carbon content to a 30 cm depth, but at present the defaults do not account for the litter layer. The IPCC guideline default for the length of period between inventories of areas for land use is 20 years, a compromise for simplicity, particularly in light of little information. This default is based on work by Davidson and Ackerman (1993), who calculate that most soil carbon loss after clearing occurs within 10 years, and work by Jenkinson (1971) which showed a buildup of soil carbon after abandonment occurs more slowly. It is also expected that response time in soils in the Tropics would occur faster than the response time in the Temperate Zone. If a soil carbon response time longer than 20 years is warranted, IPCC recommends that cohorts of areas be tracked. For example, perhaps land abandoned less than 20 years ago should be one group, and land abandoned more than 20 years ago be another group.

Comparison of Accounting Systems

The two methods–RPA and IPCC–are based on inventories of wood volumes and forest area. These sampled variables are converted to carbon estimates using relationships taken from scientific literature. The two methods can potentially produce similar estimates of carbon budgets for the United States. This is principally due to the flexibility of the IPCC method to utilize U.S. inventory data and the estimators developed by Birdsey (1992). The RPA method was based on an extensive database and developed to be specific to U.S. forests. In addition to forest identity and area, this method used age and volume to predict aboveground and belowground pools of carbon. Independent variables were chosen according to what was considered the better predictor. The IPCC method is characterized by generality and flexibility. The most general application used area and identity of land use and vegetation type to drive model predictions. Flexibility does allow for use of assumptions more appropriate for local conditions. Thus, some of the elements of the RPA method can be adopted within the IPCC framework. However, soil carbon estimates would remain largely a function of area.

Exclusions from forest carbon inventories are part of both methods. Relatively undisturbed areas are excluded from calculations under IPCC recommendations, with human disturbance as the criterion. The RPA method excludes areas characterized by low productivity and, thus, low flux per hectare. For example, the interior of the state of Alaska is excluded from the carbon calculations used by the RPA assessment. The effect of excluding lower productivity lands in the United States is estimated to alter projected inventories of carbon by less than 10 percent (see tables 4.2 and 4.3, Birdsey and Heath 1995).

Accounting for total land area is important because examining changed area in isolation will cause apparent changes in soil carbon although the changes are simply reflecting transfers between categories. Afforestation and deforestation activities affected carbon inventory simply through the movement of area in or out of forestland in the accounting of the RPA method.

Carbon inventory of these afforested and deforested lands was not counted when in the non-forest state. Because the IPCC method made a comprehensive estimate, total area remained constant. Afforestation and deforestation simply produced a transfer of area among land uses and vegetation types. Although methods differed slightly, the net effect on carbon accounting was the same for the two methods.

Previous land use is an important consideration under both systems, but different effects are assumed for the two methods. Soil carbon of afforestation is assumed to increase in the RPA analysis following regression equations developed by Birdsey, while the IPCC methodology does not include accumulation of soil carbon on these lands. IPCC methodology does include soil carbon accumulation under different land uses such as improved pasture. IPCC assumes soil carbon is constant after forest harvest and reforestation. The method used in the RPA analysis assumes that soil carbon declines by 20 percent in the 10–15 years after harvest, and then increases back to the base amount by forest age 50. IPCC methods assume a 30 percent loss of soil carbon in the 0–30 cm soil depth when temperate native soils are cultivated, with the exception of wet soils which are assumed to lose more carbon. The RPA analysis covered only the forestland, so areas of deforestation were completely removed from the analysis. Therefore, no assumptions were made about the effect of deforestation on soil carbon. IPCC recommends that cohorts of areas be tracked when soil response time is longer than 20 years, and the RPA analysis does this by tracking forest area by stand age. The discussion here focuses on qualitative implications of assumptions about trends in soil carbon in response to land use change. Further detail requires a quantitative comparison of the two methods.

The RPA accounting method does not differentiate between organic and mineral soils, nor does it identify wet soils. The IPCC method does make these distinctions. However, these distinctions are applied mostly to agricultural soils in the IPCC default methods. Ignoring this distinction probably affects the soil carbon results for the RPA method less than adopting the assumption that soil carbon accumulates under afforestation.

Recent Developments

Assumptions about trends in soil carbon following land use change can be important to carbon accounting results of both methods. Each method is designed to reflect effects of forestry practices on carbon sequestration. Thus, it is important that assumptions about soil carbon dynamics reflect scientific studies. Much of the relevant literature over the last several years has indicated that reforestation produces transient changes in soil carbon, yet other studies suggest little change occurs. In this section, we first discuss soil carbon assumptions used in the 1993 RPA analysis (Birdsey and Heath 1995) and examine how the results would change to reflect no changes in soil carbon due to harvesting. We then review current literature, focusing on the effects of harvesting, afforestation, and deforestation on soil carbon.

Soil Carbon Assumptions, Mid-1990s

The carbon analysis used in the RPA Assessment (Birdsey and Heath 1995) was based on assumptions that soil organic carbon decreased in the first 10–15 years after harvest by perhaps as much as 20 percent, with a gradual increase as the forest stand aged to maturity around age 50. At about the same time, scientific consensus leaned toward the theory that harvesting had little-to-no effect on soil carbon (Johnson 1992). Johnson (1992) concluded in his review that cultivation of forested soils results in a large loss of soil carbon, up to 50 percent in most cases, and that soil carbon usually increases substantially when non-forested land reverts to or is planted to forest. Clearing and cultivation of forests was estimated to affect soil carbon regionally by 30 percent (Davidson and Ackerman 1993), and these soil carbon changes occur mainly in the upper horizons, probably within the 0–30 cm soil depth. Deep mineral soils were seen as passive carbon pools, remaining relatively unchanging for hundreds of years.

Birdsey (1996) recalculated forest soil carbon trajectories in response to this development in the scientific literature. The model assumed no harvesting effect on soil carbon, with a 33–50% increase due to land use change to forest. For pastures becoming forested, soil carbon at age 0 was the greater of the base pasture carbon, or two-thirds of the average of the base forest carbon. For croplands, soil carbon at age 0 was taken to be the greater of the base cropland carbon, or half the average of the base forest carbon. Figure 6.4 illustrates an example of soil carbon trajectories under these revised assumptions. The effects of assumptions on soil carbon dynamics may be easily seen by comparing trends in the solid line on figure 6.3 and figure 6.4. However, note that much of the difference in absolute magnitude (as opposed to trends) between the figures is due to climatic and vegetation differences of different regions and forest types.

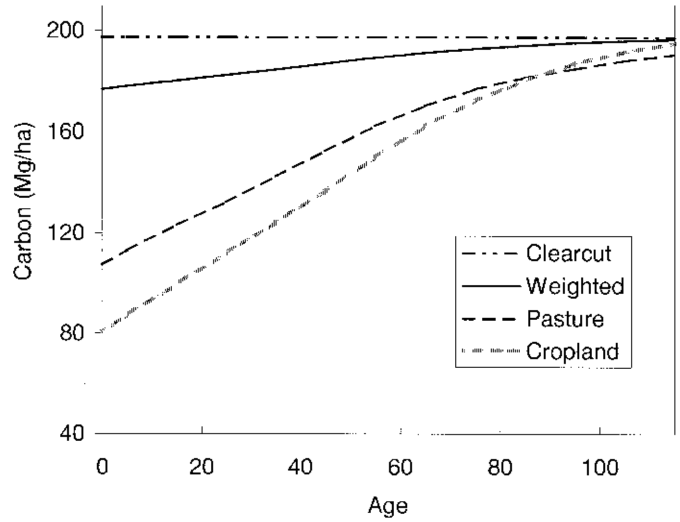

Figure 6.4—Soil organic carbon (0-1 m) by forest stand age for white-red-jack pine forest type in the northeastern United States (Birdsey 1996) under previous land use using updated assumptions. The estimates are weighted averages based on percentages of forest in previous non-forest land use.

Recent Scientific Studies on Harvesting and Reforestation Effects on Soil Carbon

Harvesting may affect soil carbon through loss of nutrients, temporary increase of slash incorporated into the soil by removal of the biomass, changes in soil physical properties such as bulk density due to physical disturbance from logging equipment, and loss of forest canopy, which affects the microclimate (Pennock and van Kessel 1997). Reforestation may act to reverse these effects. However, the actual act of regenerating the forest, including site preparation such as ripping and vegetation control, may cause soil carbon to decrease.

We are interested in changes in total soil carbon (Mg per ha). Total soil carbon is calculated by multiplying percent carbon content times volume of soil in a hectare. Davidson and Ackerman (1993) pointed out that examining percent carbon content of soil only addresses part of the carbon sequestration issue. Johnson's (1992) review, concluding that harvesting has little effect on soil carbon, was based on effects of activities on percent carbon. The number of relevant soil carbon studies has increased since 1992, and much of the results present more than simply "percent carbon." These recent studies have been more rigorously designed specifically for soil carbon. Important experimental considerations include longer duration of study, inclusion of greater amount of the soil profile, and a greater sample size that is needed to reveal significant differences under extant variability.

Clearcutting on the Hubbard Brook Experimental Forest in the northeastern United States produced a number of soil carbon changes (Johnson 1995; Johnson et al. 1995; Johnson et al. 1991; Huntington and Ryan 1990). Northern hardwoods inhabit the site, with some spruce-fir at higher elevations. Forests were logged around 1915, and there was no evidence of fire or previous cultivation. A whole-tree harvest was performed using local commercial operators; boles were removed using rubber-tired skidders. Soils were intensively sampled before, three years after, and eight years after clearcutting. Sampling intensity was such that a 10–20 percent change in total mineral soil carbon could be detected (Johnson et al. 1995). After three years, total mineral soil carbon increased 8 percent compared to pre-harvest values (which was not statistically significant at $p=0.05$). After eight years, total mineral soil carbon decreased 17 percent relative to pre-harvest values, significant at $p<0.25$. The carbon pool in the 0–10 cm layer did not differ ($p=0.92$) from pre-harvest values, but the carbon pools in the 10–20 and 20-C horizon layers decreased significantly ($p<0.05$). Percent carbon of the organic matter changed significantly ($p<0.05$) after eight years in several of the lower layers (Johnson et al. 1995), and significant ($p<0.05$) increases in bulk density were noted in the top 20 cm after three years (Johnson et al. 1995). Surprisingly, the mineral soil organic matter pool remained basically unchanged eight years after cutting (279 Mg C per ha versus 288 Mg C per ha). Thus, measuring only soil organic matter and assuming a fixed ratio of organic matter to carbon, may obscure the true carbon dynamics.

The contrast between the Hubbard Brook results and previous studies that showed no significant effects of harvesting on soil carbon was discussed by Johnson (1995). One reason proposed for the contrast was that studies may differ in sampling rules concerning disturbed areas. For instance, Covington (1981) carefully excluded disturbed sites in his study, while Federer (1984) allowed some disturbance. In this study, samples were taken across the site, regardless of soil disturbance. Huntington and Ryan (1990) reported a noticeable amount of disturbance on the site mainly due to the establishment of an extensive network of logging roads. Mixture of the top layers made delineation of the forest floor and mineral soil much more difficult and may have contributed to the 0–10 cm layer increasing in soil carbon eight years after harvest (Huntington and Ryan 1990). The apparent effect of harvesting on soil carbon in mineral soil may be more directly a function of soil disturbance at the time of harvest. Another possible reason is that the use of chronosequences in some studies may inadvertently include unknown site-specific effects.

Soil carbon dynamics qualitatively similar to those found at Hubbard Brook have also been identified in other forest types as well. Van Lear et al. (1995) combined sampling at mostly the 0–50 cm soil depth (12–20 samples at three to five permanent sampling locations at several watersheds at three depths) with modeled information at mostly the 50–100 cm depth. They studied soil carbon dynamics after harvest of a 55-year-old loblolly pine forest on an eroded, previously cultivated site in the Piedmont of the southeastern United States. They found an increasing soil carbon trend after harvest, which quickly decreased below pre-harvest carbon, but by 13 years after harvest it had increased above pre-harvest soil carbon levels. This is illustrated in figure 6.5, which somewhat resembles the older accepted theory of soil carbon dynamics shown in figure 6.1. However, some of their information in the 50–100 cm horizon was estimated, not measured.

Pennock and van Kessel (1997) conducted a study on chronosequences to examine the effects of clearcutting in six aspen-white spruce stands in central Saskatchewan, Canada. They sampled from 0 to 45 cm. Results showed a significant ($p<0.05$) increase of 8 percent in soil carbon less than five years after the clearcut, with a significant decrease of 23 percent 6–20 years after clearcutting as compared to mature forests. However, one caution in interpreting the results is that they did not separate the forest floor from the mineral soil surface on clearcut sites that had been prepared for tree planting. The surface layer was missing or very thin on these sites, and it was felt that measuring the forest floor separately would have introduced more error than measuring it with the mineral soil.

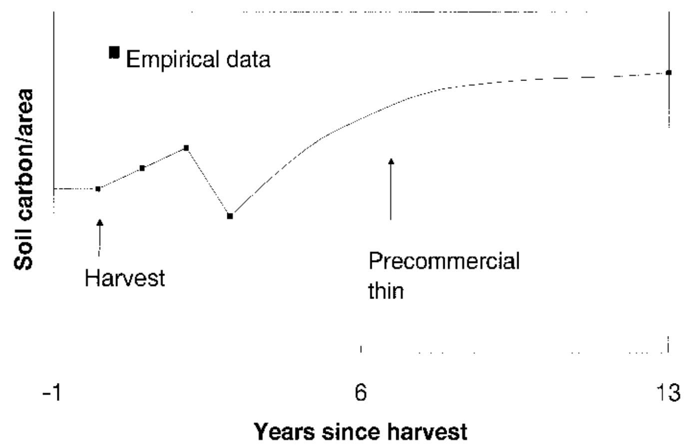

Figure 6.5—Trend of soil organic carbon after pre-harvest low intensity burns, and clearcut harvest of a 55-year-old loblolly pine stand on eroded soil (following Van Lear et al. 1995).

Sampling from a chronosequence suggests an increase in soil carbon with reforestation, even where the transitory increase immediately after harvest did not appear. Using a chronosequence, Entry and Emmingham (1995) found an increase in soil carbon in the top 10 cm of Douglas fir stands of increasing age. The stands were categorized as young-growth (about 30 years old), second-growth (about 66 years old), and old-growth (from 120 to 300 years old). Soil carbon almost doubled between young-growth and second-growth but the increase was not significant at $p<0.05$. Soil carbon of the old growth was almost three times that of young-growth, and it was significantly greater ($p<0.05$) than both younger groups.

Strong (1997) studied five cutting treatments with three replications each in northern hardwoods in northeastern Wisconsin using a chronosequence (pre-treatment values were not known for each treatment). The forests were logged in the early 1900s, and were generally even-aged. The study was initiated in 1952 and has been continuing for 40 years. The treatments included replications of control, diameter-limit cut, and three levels of individual tree selection. No trees were cut in the control, all trees 20.3 cm and larger were cut in 1952 in the diameter-limit cut, and heavy, medium, and light individual tree selection was performed in 1952, 1962, 1972, and 1982. There was no statistical difference ($p<0.05$) in soil carbon in the 0–40 cm horizon, but there was a significant difference in the 3–10 cm depth, and there was a trend of increasing soil carbon as basal area increased. This implies that soil carbon may decrease with increasing harvest intensity. Unfortunately, because this study uses a chronosequence, it may be that the soil carbon differences between treatments are due to initial site differences, not to harvesting intensity.

Other studies were reviewed but may be of limited usefulness. We reworked the data in Frazer, McColl, and Powers (1990) and found soil carbon increased 13 percent by five years after clearcutting, and decreased 11 percent by 18 years after clearcutting under regeneration, relative to an uncut area. However, there were no replications and therefore no statistics, and they sampled only to a depth of 14 cm. Olsson et al. (1996) studied soil carbon in Scots pine and Norway spruce at four sites in Sweden. They examined the top 0–20 cm some 15–16 years following harvest, and bulk density was assumed to remain constant in their calculation of total soil carbon. The three types of harvests studied were conventional stem harvest (residues left on site), harvesting all aboveground tree parts except needles, and whole tree harvesting (no residues left on site). If we assume harvesting did not impact bulk density, soil carbon increased fairly consistently in the 0–20 cm layer by about 5 Mg C per ha (not statistically significant at $p<0.05$) on each site. There were no significant trends of harvest intensity over all sites; however, disturbance from harvesting was carefully avoided. Operators tried to avoid soil compaction and mixing of soil layers. Black and Harden (1995) studied the effect of clearcutting in a mixed conifer stand in California. They sampled soil in the 0–20 cm layer in stands of six different ages but found no strongly consistent trend. They did note the younger stands (0–79 years old) in general contained more soil carbon than the old-growth stand. They concluded that other factors besides harvesting confound results. We also reviewed other studies, but we decided not to include them because they included only percent carbon, or were limited in duration or in depth (for instance, Knoepp and Swank 1997).

Afforestation and Deforestation

We review afforestation and deforestation studies together, because soils in U.S. forests are generally accepted to lose soil carbon when cleared and cultivated and then accumulate soil carbon after the land is revegetated with forest. Forest soil eventually accumulates carbon to a maximum level regardless of previous land use, unless severe erosion has occurred. Similarly, expected decreases in soil carbon partly depend on use after deforestation such as annual crops, pasture land, or urban development. For example, as mentioned previously Davidson and Ackerman (1993) conclude soil carbon decreases regionally by 30 percent (ranging from 20 to 40 percent) in the entire soil column when forests are cleared and the land cultivated.

The 30 percent loss is generally accepted as the magnitude of soil carbon change for deforestation and cultivation; however, the length of time over which the loss takes place is still being debated. How long does it take to reaccumulate soil carbon to its "maximum" level? Huntington (1995) studied a chronosequence of sites in the Piedmont of the southeastern United States, measuring soil carbon to the depth of 1 m for some sites. An upper bound for soil carbon for a cultivated, eroded forest soil was estimated using current soil carbon in the 50–100 cm layer, and the lower bound was estimated using soil carbon in similar soils currently under cultivation in the area. Using these upper and lower bounds, Huntington (1995) estimates a range within which soil carbon accumulates after forest regeneration on these lands. His results are shown in figure 6.6. He estimated an increase of 0.34 to 0.79 Mg C per ha per year (34–103 percent) accumulated over 70 years. Most soil carbon was lost within the first 35 years following clearing and cultivation. Van Lear et al. (1995), working on the Piedmont in South Carolina, estimated an increase of 0.47 Mg C per ha per year (220 percent) over a period of 55 years. This percentage is high because of the low initial soil carbon content of the site. Schiffman and Johnson (1989) estimated about 0.50 Mg C per ha per year (about 35–57 percent) accumulated on eroded soils in Virginia. Eroded soils present a special problem for accounting because eroded soil carbon may not decompose and be released to the atmosphere. It may be deposited elsewhere as soil carbon.

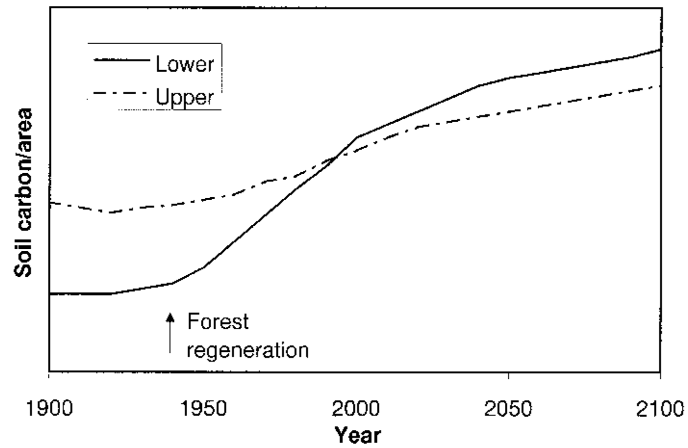

Figure 6.6—Upper and lower range of dynamics of soil carbon after forest regeneration on eroded cultivated soil in the southern United States. (Adapted from Huntington 1995.)

Trends of Soil Carbon in Current Literature

Based on this preliminary review, soil carbon dynamics following harvest appear to depend on the amount of disturbance caused by logging operations. The disturbance associated with some commercial harvests may cause soil carbon to increase initially in the first few years by 8–13 percent, then decline to below initial values by 11–20 percent by 10–20 years after harvest, and eventually increase again. Some studies showed changes in soil carbon below the 0–30 cm depth, indicating that experimental soil studies should sample lower soil depths. Severely eroded soils also create additional problems concerning depth because much of the original soil may be eroded. Results compared at apparently equivalent depths in eroded and non-eroded soils may be difficult to interpret. Other aspects of sampling designs identified as potential problems include initial site differences in the use of chronosequences, use of percent carbon as a proxy for total carbon, and the need for appropriate sample size to produce significant results. Soil carbon following deforestation and cultivation declines about 30 percent in the entire soil column within 30 years of cultivation. Soil carbon increases gradually following afforestation with good stocking, increasing by 30 percent at a rate more gradual than the decline following deforestation.

Summary

We reviewed two accounting systems, one developed by the IPCC (1997) and the other from the 1993 RPA assessment (Birdsey and Heath 1995). Both systems base predictions on the forest inventory variables of volume and area. Both methods recognize the importance of previous land use. The IPCC default system explicitly counts soil carbon loss when forests are cleared and cultivated but does not include the accumulation of soil carbon due to afforestation, although soil carbon increases due to differing agricultural tillage practices are included. Birdsey and Heath (1995) explicitly account for the accumulation of soil carbon due to afforestation but do not explicitly count soil loss after deforestation. This is because the RPA analysis focused only on carbon in the forest sector. Deforested areas were assumed counted in the agricultural or urban sector, not forests, and over the last 30–40 years more land has become afforested than deforested.

Recent scientific studies indicate that harvesting may influence soil carbon, an initial slight increase followed by a decrease, and finally an increase. We speculate that soil carbon will eventually return to pre-harvest levels. This corresponds to the pattern in the soil carbon assumptions in the RPA analysis. The magnitude of the effect seems to depend on the level and type of disturbance from logging operations. Countries with active forest management, such as the United States, should give further consideration to the overall level of disturbance in harvesting operations and revise soil carbon assumptions accordingly. Soil carbon decreases for 20–30 years following deforestation and cultivation and then remains relatively constant; following afforestation, soil carbon increases at a more gradual rate than the rate at which it had decreased, eventually becoming somewhat stable.

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CHAPTER 7

Considerations for Interpreting Probabilistic Estimates of Uncertainty of Forest Carbon

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Introduction

Quantitative estimates of carbon inventories are needed as part of nationwide attempts to reduce net release of greenhouse gases and the associated climate forcing. Naturally, an appreciable amount of uncertainty is inherent in such large-scale assessments, especially since both science and policy issues are still evolving (Brown and Adger 1994; Klabbers et al. 1996; IPCC/OECD/IEA 1997a). Decision makers need an idea of the uncertainty in carbon estimates in order to consider tradeoffs between known effects, possible outcomes, and preferred consequences. While an ultimate goal of assessments is to minimize uncertainty, a more immediate concern is to adequately quantify existing uncertainty. The goal of this chapter is to present some useful considerations for the interpretation and subsequent use of information from probabilistic assessments of uncertainty.

Forests store a large portion of the carbon in terrestrial ecosystems; therefore the extensive and largely managed timberlands of the United States represent a potential for producing offsets to carbon dioxide emissions (Birdsey 1992; Heath et al. 1996; Sohngren and Haynes 1997). Carbon content is a function of the state of forests: size, age, composition, productivity, and area, for example. These, in turn, are dependent on histories of management, utilization, weather, disturbance, and land use. Finally, all of these variables can be manipulated in many ways to fit differing scientific modeling approaches, as demonstrated by other chapters in this volume and citations therein. Decision makers faced with such complexity are likely to want information about uncertainty.

Uncertainty is a natural element of scientific understanding and therefore also an element of simulation modeling. This is the case for many forest-system models where uncertainty is sometimes explicitly quantified, sometimes disregarded, but most often discussed in general qualitative terms. Uncertainty in models is sometimes poorly characterized because the primary purposes of many models are to present best estimates or evaluate cause-and-effect relationships, not emphasize what is unknown. Additionally, "uncertainty" itself is sometimes a poorly defined, or elusive, quantity (Morgan and Henrion 1990; Shackley and Wynne 1996). A complete quantitative estimate of total uncertainty in forest carbon budget projections is beyond the scope of this chapter. Fortunately, models of uncertainty are useful, even when they do not provide a "bottom line" (Morgan and Henrion 1990; Cullen and Frey 1999).

Decisionmakers, or anyone using quantitative assessments of uncertainty, will likely face the need for pooling, comparing, or otherwise synthesizing such assessments. Because such actions are essentially modeling, some understanding of the process may be beneficial. Here, we particularly emphasize the consequences of summarizing uncertainty, as well as how such summaries can affect the perception of uncertainty in subsequent use of the information. Our discussion is oriented toward providing decision makers with an overview of some links between the form assigned to uncertainty and the perception of that uncertainty. Examples are presented from our current forest carbon budget modeling efforts where we employ probabilistic definitions of uncertainty in Monte Carlo simulations. The method of summarizing model results can affect perceived uncertainty, and summing uncertainty without considering covariability among parts can create a false estimate of uncertainty. Details on methods of analysis are in Smith and Heath, (in press) and data are summarized from Heath and Smith (2000).

A Forest Carbon Budget Model: FORCARB

The model FORCARB was developed to estimate carbon budgets for U.S. forests (Heath and Birdsey 1993; Plantinga and Birdsey 1993; Birdsey and Heath 1995; Heath et al. 1996). Carbon budgets, as used here, are essentially estimates of size for various pools of carbon inventory as well as net changes over time. Net change in carbon inventory is referred to as flux. FORCARB is linked to a system of models (Mills and Haynes 1995; Birdsey and Heath 1995) developed as part of the periodic Resources Planning Act timber assessments (Haynes et al. 1995). Inputs to FORCARB from other models include landscape-scale projections of age-structure, volume, and area (Mills and Kincaid 1992), and as such, they implicitly contain a wide array of uncertainties. The focus in these simulations was on uncertainty within the FORCARB model, thus all inputs from other models were assumed known with certainty.

Functional relationships are used to estimate carbon pool sizes for hardwood trees, softwood trees, understory, forest floor, and soil based principally on age and volume inputs. An example of such a relationship is shown as the solid line in figure 7.1. Here, forest floor carbon inventory is estimated from stand age. Subsequent reference to a "FORCARB parameter" refers to this type of functional relationship. Carbon pools are then expanded to total carbon for large areas of similar forest-type and productivity within a region. These large areas are termed "forest management units" (10^3 to 10^7 ha with a median of 180,000 ha for the 1990 inventory). Regional subtotals are formed and, finally, summed to a national total. Private timberlands in the 48 contiguous states are represented by results presented here, which include 216 forest management units. Carbon budget projections are presented in greater detail in Heath and Smith (2000). The basic sequence of a FORCARB simulation is illustrated in figure 7.2.

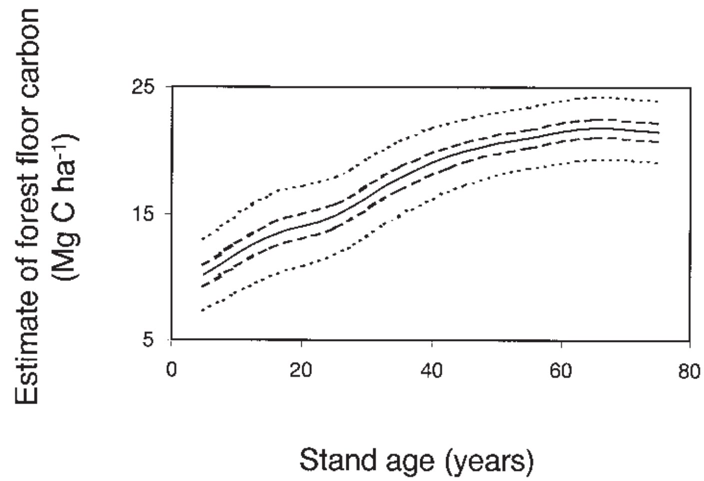

Figure 7.1—An example of a typical functional relationship (or FORCARB model parameter) used to project forest floor carbon inventory based on stand age (solid line). Probability bands illustrate our meaning and use of uncertainty in "FORCARB parameters" for this analysis. The bands indicate the 5th, 25th, 50th (expected value), 75th, and 95th percentiles (bottom to top respectively) of the probability distribution around the dependent variable. (Relationship is from a Douglas fir forest management unit.)

Uncertainty

Some level of uncertainty is usually a part of any model, assessment, or decisionmaking whether or not it is an explicitly considered part of the process. A widely used and potentially general term such as "uncertainty" can be confusing or misleading unless it is adequately defined (Hattis and Burmaster 1994; Shackley and Wynne 1996). At its simplest level, uncertainty can be the state of not knowing, or the inability to quantify something with a single discrete value. Sources of uncertainty can vary widely, and as a consequence, attempts to narrow the definition can require reference to variability, ignorance, systematic error, unknowns, expert opinion, semantics, or misapplication of a model (Morgan and Henrion 1990; Hattis and Burmaster 1994; Rowe 1994; Ferson and Ginzburg 1996; Cullen and Frey 1999). In earlier literature (largely stemming from Knight 1921), scientists were careful to define the risk of an event by a probability based on documented frequencies of occurrence. Risk was contrasted with uncertainty where such probabilities could not be assigned. However, current applications employ a range of definitions for uncertainty, including probability; furthermore, valid definitions of probabilities can include observed frequency or even subjective expectation (Hoffman and Hammonds 1994; Reckhow 1994; Dakins et al. 1996; Schimmelpfennig 1996; Paoli and Bass 1997; Haynes and Cleaves 1999). We use a probabilistic definition of uncertainty.

An unknown, but unique, inventory of carbon exists within a given forest management unit for a particular year. Our inability to precisely specify that value is the general definition of uncertainty we employ here. This concept of uncertainty implies that we can specify a range of possible values and an associated likelihood for values within that range. This describes a probability distribution, or more properly, a probability density function (PDF). Thus, we use PDFs as convenient quantitative and graphical representations of uncertainty (Vose 1996; Cullen and Frey 1999).

The effect of this definition of uncertainty, applied to estimating carbon for a given subset of a forest management unit, is illustrated in figure 7.1. The broken lines are probability bands indicating specific points on dependent variable PDFs–or uncertainties–about exact values of carbon per unit area. These probabilities reflect uncertainty in predicting carbon from stand age. Normally distributed PDFs were assumed to describe uncertainty about FORCARB parameters (details in Heath and Smith, 2000). No assumption of normality was required for this model: its use was simply a convenience for describing assumed expected values with symmetrical distributions. Analyses would ideally address all sources of uncertainty relevant to policymakers' questions about forest carbon inventory and flux. However, as mentioned above, a pragmatic first step is to focus on uncertainty internal to FORCARB. Therefore, uncertainties presented here are limited to this portion of the potentially much larger system of models that describe forests.

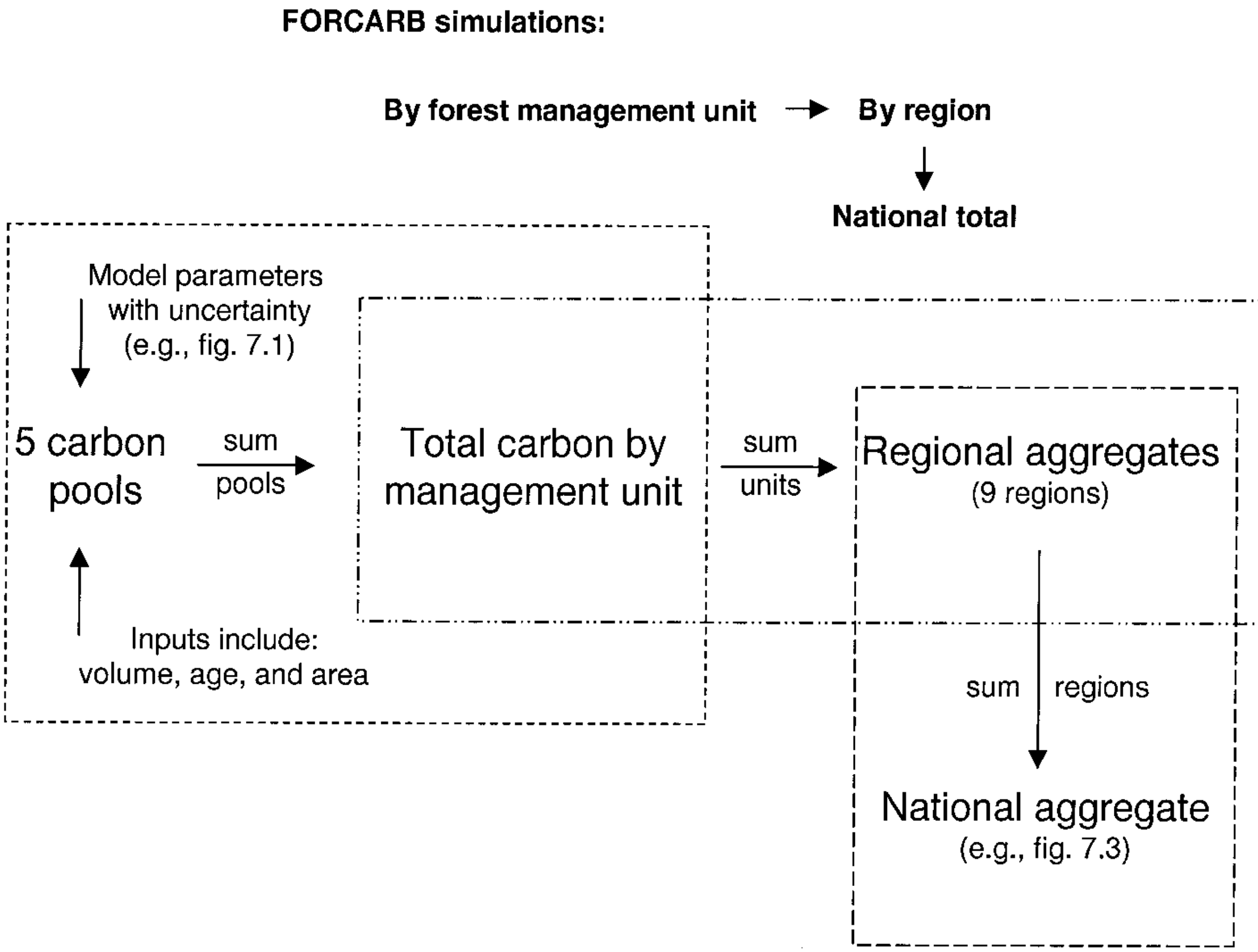

Figure 7.2—Graphic depicting organization of FORCARB simulations to estimate carbon inventory for individual forest management units (leftmost box), regional subtotals (upper right), and the national total (lower right). FORCARB estimated five carbon pools that were summed for total carbon inventory per forest management unit. A total of 216 such simulations were made for the national total.

Method of Simulating Uncertainty

An uncertainty analysis is a modeling process that is implemented for two related purposes–estimating uncertainty and identifying influences on that uncertainty (Morgan and Henrion 1990; Cullen and Frey 1999). We estimate uncertainty in the FORCARB model by employing Monte Carlo simulations with Latin Hypercube sampling (Morgan and Henrion 1990; Vose 1996; Cullen and Frey 1999). This is but one of a number of approaches to uncertainty analysis, and we apply the method here to estimate uncertainty in forest carbon budgets.

A Monte Carlo simulation is produced through repeating a basic model simulation for a large number of iterations. One value is randomly selected from each input PDF for each iteration. For example, random selection from a PDF describing the parameterized relationship shown in figure 7.1 would produce estimates of forest floor carbon inventories ranging between approximately 10 and 16 Mg C per ha for 15-year-old stands. A different single value would be randomly selected for each iteration of the Monte Carlo simulation with most selections being near 13 Mg C per ha. Each iteration produces a single-valued model result. An accumulation of many such individual results produces a distribution representing the results of the Monte Carlo simulation. Latin Hypercube sampling is simply a stratified sampling procedure in which distributions are sampled from equal-probable intervals, without replacement, thus reducing the sampling required to fully represent PDFs. The number of iterations included in a simulation affects precision of resulting distributions. Results provided here were from 100 iterations, which were adequate to define the shape of distributions for the quantities we examined.

We employ Monte Carlo simulation for uncertainty analysis because it features four principal advantages: 1) expressions of likelihood; 2) analysis of influences; 3) flexibility; and 4) explicit representation of covariability among parts (Morgan and Henrion 1990; Joint Climate Project 1992; Dakins et al. 1996; Morgan and Dowlatabadi 1996; Vose 1996; Cullen and Frey 1999). Although a first question often asked about uncertainty concerns identification of possible extreme events, this can quickly lead to a need to identify the likelihood of specific events between the extremes. Results as PDFs specify the range of possible outcomes together with their respective probabilities—both central tendencies and extreme events. The second factor is an advantage because influences on results are usually not evenly distributed among the components of a model. Identifying most-influential components as they affect overall uncertainty or even a tendency toward extreme results has utility for both model developers and policy analysts. Third, questions asked of an analysis are likely to change, and the same is true for information going into an analysis. This is a simple and flexible approach relatively free of restrictive assumptions. For example, although normal distributions were input for model parameters as a convenience, there were no required assumptions about distributions nor any need to know central moments. Finally, Monte Carlo simulation explicitly accounts for covariability among all derived PDFs. The third and fourth characteristics are of most interest here: minimal assumptions and explicit representation of relatedness among parts of the model.

Results and Discussion

Values for carbon budgets and uncertainty presented here are based on results of Heath and Smith (2000) and represent preliminary estimates for private timberlands. This chapter is intended to illustrate links between summary values extracted from PDFs and the perception of uncertainty associated with use of the summaries. Results are presented in three parts. First, we discuss considerations for avoiding the loss of important information when forming tabular summaries of PDFs. These results underscore the usefulness of need-specific summaries and careful definition of terms so that summaries reflect the interests of users. Second, we discuss additional considerations necessary when combining a number of PDFs. Here, disparity in size and dependencies (or covariability) among PDFs become important. Finally, we discuss some implications of these results for expanding the uncertainty analysis to the larger system of models.

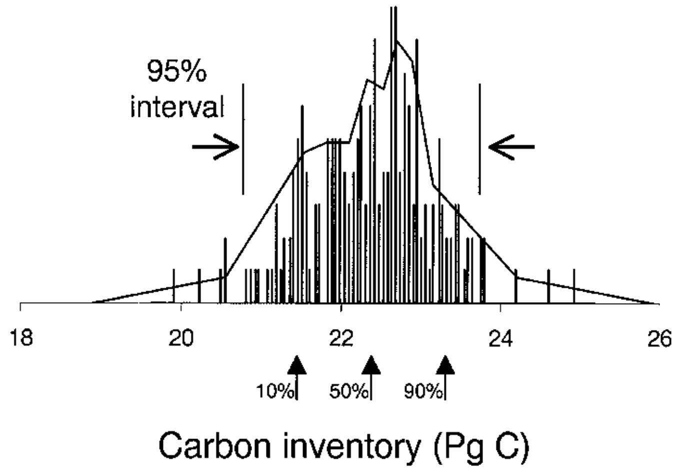

Figure 7.3—Estimate of carbon inventory (billion metric tons) of private timberlands for 2000. Model results presented as a histogram and smoothed probability density produced by Monte Carlo simulation. The central 95 percent of the distribution may be considered analogous to a 95 percent confidence interval. Arrows indicate carbon levels for the 10th, 50th, and 90th percentiles, commonly used to summarize low, median, and high simulation results, respectively.

Tabular Summaries from Continuous Distributions

Frequency distributions of model results are initial products of Monte Carlo simulations. A result of uncertainty in FORCARB projections of carbon on private timberlands for the year 2000 is shown in figure 7.3. The figure shows both a histogram of individual results from the many iterations of the Monte Carlo simulation and the smoothed distribution fit to the histogram. PDFs are formed from frequency distributions by normalizing the distribution, or setting the total area under the smoothed histogram to equal one (the cumulative probability of all values).

Probability densities are easily interpretable graphics of quantitative expressions of uncertainty. The likelihood that total carbon inventory will be within a given range, for example, is in proportion to the appropriate area under the PDF. Graphical presentations facilitate quick comparisons among a few such expressions of uncertainty, and numerical comparisons among whole distributions are similarly possible. However, interest in uncertainties in integrated assessments can often focus on specific values such as thresholds or ranges. As such, summarizing PDFs using a few numbers is often desirable when integrating large amounts of information.

Uncertainty represented in tabular form is usually presented as either individual points or an interval along the PDF. Such summaries do not convey the exact shape of the distribution, but they do reduce discussion to a few key values. The use of individual points is shown in the carbon budget summary presented in table 7.1. A percentile indicates the portion of the PDF less than the given value; this can also be interpreted as the probability of results less than or equal to that value. For example, uncertainty in the model suggests that carbon inventory in 2000 will be less than 23.3 billion metric tons (Pg) with a probability of 0.90 (table 7.1 and fig. 7.3). Distribution percentiles such as the 10^{th}, 50^{th}, and 90^{th} are commonly used to summarize low, median, and high simulation outputs, respectively. Intervals can be based on select percentiles (10^{th} to 90^{th} percentiles, for example) or formed around a central value such as the mean or median. Intervals around a central value can be expressed as relative or absolute values. For example, a symmetrical interval about the median carbon inventory in 2000 can be given as ±10 percent or ±2.2 Pg C–relative or absolute, respectively.

Tabular representations of uncertainty can be useful simplifications of results from uncertainty analyses. How uncertainty is summarized and presented should reflect the key features necessary for subsequent use of the information. There are two somewhat obvious, but important, caveats to note when using tabular summaries of uncertainties. The first is the link between the shape of the distribution and the interval. Selection of either interval or level of confidence determines the value of the other without reference to properties of a standard distributional form (also known as a parametric PDF, such a normal or lognormal, for example). An implication of this is that the interval of ±1 standard deviation about the mean does not necessarily enclose 68 percent of the distribution as would be the case under an assumption of normality for a PDF. However, a PDF obtained through Monte Carlo simulation can be represented by a close equivalent parametric PDF with the amount of information lost proportional to the closeness of the fit. The importance of such a compromise depends on the information represented by the PDF and its subsequent use. The second consideration is the distinction between representing uncertainty as a relative or an absolute interval. Both are reasonable representations of uncertainty, yet the dual definitions can be a source of confusion when making comparisons. The same absolute average range when applied to different median values can produce very different relative ranges. For example, the approximate ±4 percent of median inventory given in table 7.1 represents a considerably larger amount of carbon than the approximate ±15 percent of median

Table 7.1—Estimated total carbon inventory for private timberlands for 1990 and 2000, and average annual net carbon flux for the interval. Values are from the 10^{th}, 50^{th}, and 90^{th} percentiles of the respective probability densities produced through Monte Carlo simulation. Positive flux indicates that carbon is being sequestered in the forest. Tg =1 million metric tons.

	Total inventory (Pg C)		Flux (Tg C y^{-1})
Percentile	1990	2000	1990–2000
10	20.7	21.4	63
50	21.7	22.4	74
90	22.6	23.3	86

flux. Simple and clear definition of how uncertainty is summarized can eliminate most confusion.

Choice of interval (or subset of PDF) to represent uncertainty presumably depends on the needs of the individual user. Here, an expression of confidence is simply the summed probability along this interval, obtained directly from the distribution. The relationship between an interval and confidence is determined by the shape of the probability distribution. These ideas are illustrated by figure 7.3. The interval analogous to the 95 percent confidence interval is between the 2.5 percentile and the 97.5 percentile (p([20.8,23.7])=0.95). This same interval can be expressed as averages of ±7 percent or ±1.5 Pg C around a median value of 22.4 Pg C. Here, the choice of a 95 percent level of confidence (probability=0.95) implicitly determined the size of the interval. Similarly, the choice of an interval, such as ±5 percent of the median, is simply the reverse of this process. Plus or minus five percent of the median value (1.1 Pg C) comprises about 86 percent of the distribution (p([21.3,23.5])=0.86). Note that the "plus or minus" values we present are averages of the two intervals for the nearly-symmetrical distributions, and methods of establishing confidence intervals vary among applications (Morgan and Henrion 1990; Cullen and Frey 1999).

A single example can usefully reiterate the ideas presented in the two preceding paragraphs. Simply stating that a level of uncertainty is ±10 percent: 1) ignores much of the information from a PDF such as change in expectation across that range; 2) implies that uncertainty is strictly a function of the size of the expected value; and 3) says nothing about confidence in the range provided. Level of ambiguity in specifying uncertainty does not imply any level of "correctness" for an analysis, but it can influence confidence. Simply put, tabular summaries, even "±10 percent," can be entirely appropriate; however, the key issue is information provided or lost. Understanding both the information needed and the information available can lead to informed choices about tradeoffs.

The benefit of summarizing PDFs should exceed the relative cost of lost information.

Summing 216 Forest Management Units for an Aggregate Total Uncertainty

Results from the carbon budget model presented here are aggregate uncertainties that represent the sum of PDFs from 216 forest management units. While our examples are taken from a simulation model, decisionmakers are likely to face similar considerations with multiple PDFs. Information is commonly acquired from a number of separate sources, and this can present the need for comparing or summing a number of results. Therefore, considering relatedness among PDFs is an appropriate addition to a discussion of PDF summaries. The simplest procedure for summarizing and summing many PDFs is probably through application of the central limit theorem (Morgan and Henrion 1990; Cullen and Frey 1999). This assumes relatively balanced contributions among each of the PDFs summed and independence among PDFs. Under these conditions, the sum is expected to be normally distributed, and the variance of the sum is equal to the sum of the variances.

Disparity among size of the 216 forest management unit carbon inventory pools can influence control over total carbon and total uncertainty. If most of the total carbon inventory is attributable to a few large forest units, then research to improve the parameter estimates of these units will usually contribute more to improve estimates of total carbon inventory than improving the parameters of smaller forest management units. The larger 12 percent of the private timberland units simulated for this study account for more than two-thirds of the total carbon (fig. 7.4). That is, only 12 percent of the management units exceed 0.2 Pg C (the second size class in fig. 7.4), yet they account for over two-thirds of the total C inventory. The uncertainty of parameters of the smaller units would have to be extremely large to produce greater absolute uncertainty than the large units. The disparity in size among the 216 forest management units suggests that the PDF of an aggregate total could not be determined through simple application of the central limit theorem.

Determination of independence, or conversely dependence, among PDFs depends on both prior knowledge of the values and the modeling process. The meaning assigned to uncertainty of input PDFs, or FORCARB parameters, becomes critical as the separate pools are summed. We use uncertainty as an expression of our expected level of ignorance. For example, uncertainty includes our inability to translate an independent variable such as an exact volume of timber on an exactly specified area of land to a precise quantity of carbon in the system. If our ability to make that estimate is similar across forest types and regions then the estimates of uncertainty would be jointly related or highly correlated. However, as the estimates become more dependent on elements of biology, management, ecology, or biogeochemistry of the respective forests, the degree of independence among the separate estimates will tend to increase. Similarly, if we view uncertainty as simply random variability, then the separate estimates made for different forest types would also be considered independent.

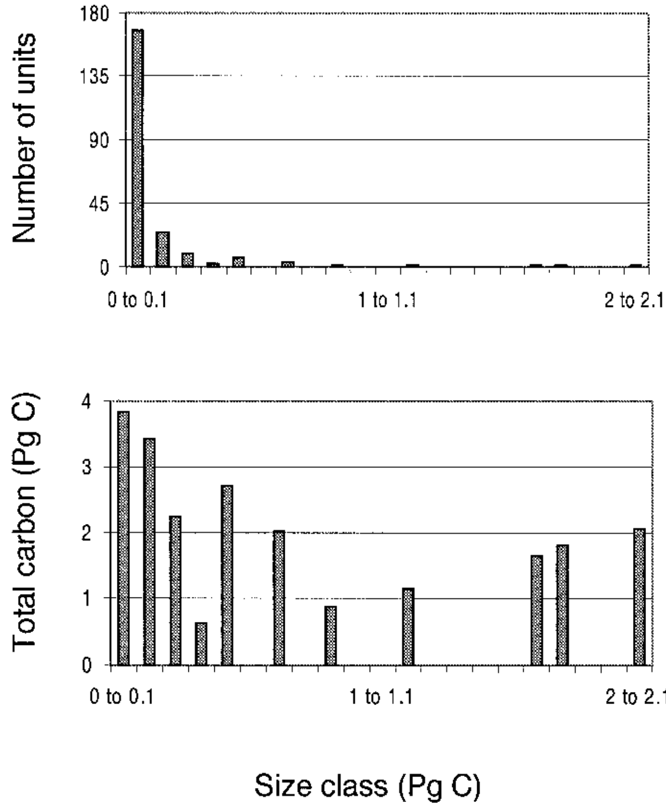

Figure 7.4—Histograms illustrating the disparity in size of carbon inventories among the 216 forest management units contributing to the national estimate for the year 2000, in terms of (a) number of forest management units per size category and (b) total carbon (sum of units) per size category (billion metric tons).

Assumptions about covariability among 216 separately determined forest carbon pools can have a tremendous effect on the apparent uncertainty of the total. FORCARB simulations in Heath and Smith (2000) reflected a relatively high degree of joint correlation–generally with coefficients of correlation between 0.60 and 0.98. Figure 7.5 shows the possible effects of covariability among the forest management units. The 216 distributions were specified as having joint correlations with coefficients of correlation of approximately 0.05, 0.50, and 0.95 (low to high covariability) based on modifying their rank orders from the Monte Carlo simulation (Iman and Conover 1982). This was simply a numerical manipulation to dem-

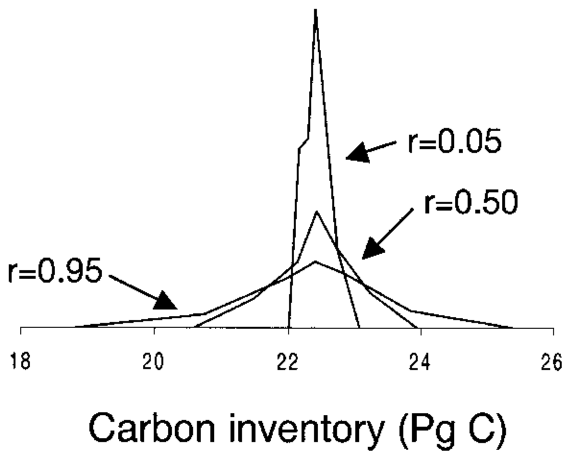

Figure 7.5—Hypothetical estimate of carbon inventory (billion metric tons) of private timberlands for the year 2000 as affected by covariability among PDFs for each of the 216 forest management units. Before summing the separate PDFs, correlation coefficients were set at approximately 0.05, 0.50, and 0.95 to produce narrowest to widest distributions for the total, respectively.

onstrate the effect of covariability on apparent uncertainty. The probabilities of high-valued samples are largely canceled out by low-valued samples when the summed distributions are considered largely independent (with coefficient of correlation, r=0.05). This central tendency produces a relatively narrow distribution in contrast to high correlation where factors leading to higher-valued samples of carbon in one system would also lead to higher-valued samples in another. The interval between the 10th and 90th percentiles was 4.5 times greater with r=0.95 than with r=0.05. We emphasize that manipulations done here were simply a means of demonstrating the consequences of covariance terms and the importance of any assumption about independence.

Average annual carbon flux is based on the difference between PDFs representing carbon inventory estimates (fig. 7.6). Here too, the value of the covariance term is important. With independence between the two inventories, uncertainty of the flux estimate is directly proportional to uncertainty in the two distributions. However, non-zero covariance affects the size of the flux PDF, as illustrated in figure 7.6 by manipulations of the coefficient of correlation between the two inventory PDFs. In general, range of uncertainty in estimated average annual flux

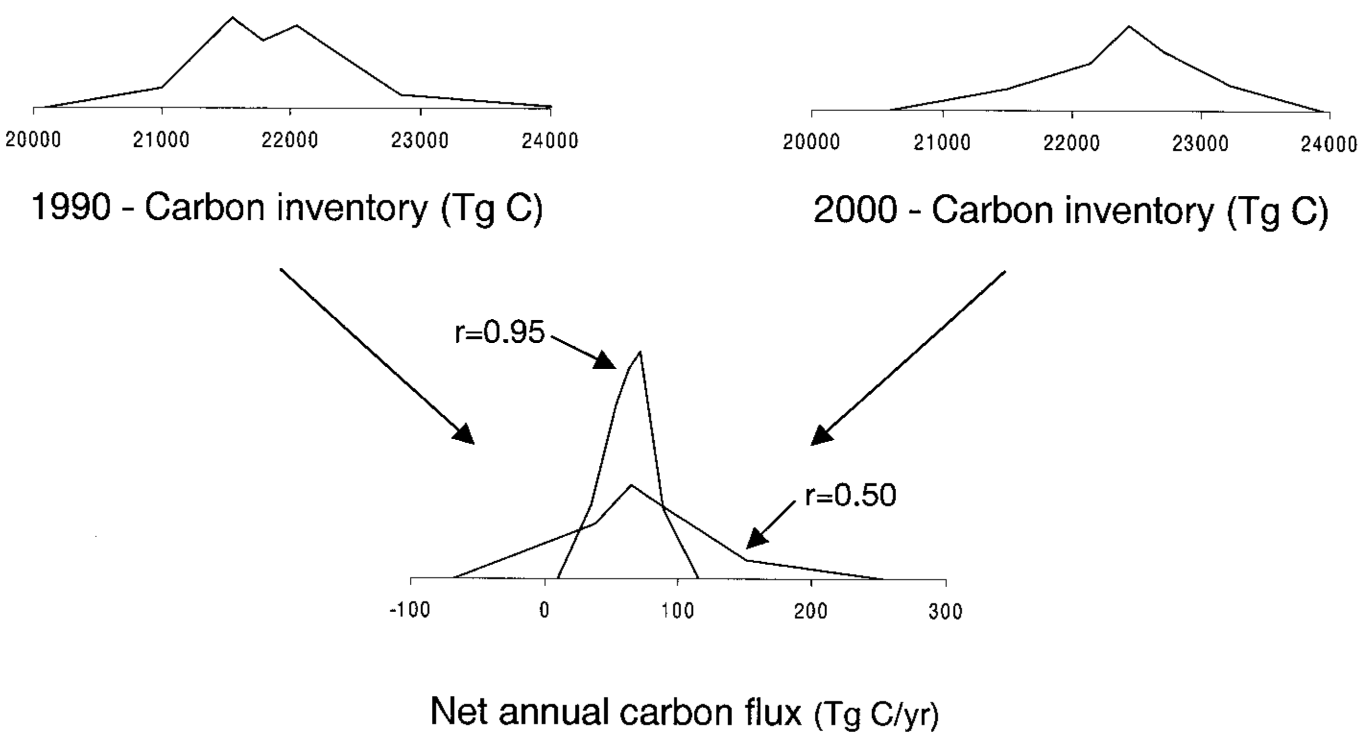

Figure 7.6—Examples of the effects of covariability between estimates of carbon inventory (million metric tons) on average annual net flux (million metric tons per year) uncertainty. Estimates for carbon inventory of private timberlands for years 1990 and 2000 were based on joint correlations among forest units set at r=0.5. Hypothetical average annual flux PDFs were calculated using correlation coefficients between years set at 0.50 and 0.95, producing wide and narrow distributions, respectively. Flux calculations were based on annualized difference between 1990 and 2000 distributions. Positive flux indicates that carbon is being sequestered in the forest.

is inversely proportional to covariance between inventories. If similar information was used to estimate carbon in each of the two years then the two distributions would be highly correlated. This was the case here (table 7.1) where age and volume were specified without uncertainty and FORCARB model parameters (similarly applied in each year's estimate) were the only sources of uncertainty.

Sums and differences of related PDFs depend on addition and subtraction of covariance terms, respectively. These are straightforward calculations if complete variance-covariance tables are readily available. Such information may be provided with original data sets, or it can be explicitly simulated within models. However, full knowledge of covariances is not a very realistic expectation when facing separately acquired estimates of uncertainty from independent sources. Nevertheless, even simple qualitative information can be usefully applied to sorting through post-analysis PDFs. For example, simply knowing that some positive, but unspecified, level of correlation exists between a pair of variables would lead an analyst to place more confidence in summaries where values were jointly drawn from similar regions of the respective PDFs. Another example of information provided by even limited knowledge of covariability is the effect of uncertainty in two inventory PDFs on uncertainty in estimated flux. The assumption of independence between inventories is a conservative assumption leading to large uncertainty in flux. Any knowledge of relatedness between the two inventories will reduce flux uncertainty, even without reducing uncertainty of the respective inventory PDFs.

Implication for a Larger External System

Decisionmakers are seldom provided probabilistic estimates of uncertainty without any accompanying information applicable to its use or context. Similarly, they are unlikely to be faced with summing 216 separate PDFs. The modeling examples were provided here to illustrate considerations for summarizing PDFs as descriptions of uncertainty. The effects of tabular summaries and relatedness are also useful when addressing issues of many uncertainties in a complex system.

The system defined by the FORCARB model is clearly a subset of a larger integrated system. Concern over the prospect of rapidly growing uncertainties as more elements are brought into an analysis cannot be quantitatively addressed without comprehensive uncertainty analyses. However, the results provided here do illustrate: 1) the effects of covariability among parts; and 2) how the definition of an interval affects the perception of uncertainty. For example, the interval between the 10th and the 90th percentiles of the 1990 carbon inventory PDF for the Northeastern Forest Industry Maple-Beech-Birch forest management unit (result not shown) is about 10 percent of the median. The corresponding interval for the national total, after adding the additional 215 forest management units, is only about 9 percent of the median (table 7.1). The same interval could range from 3 and 12 percent of the median by simply adopting different assumptions about covariability among forest management units as illustrated in figure 7.5. Relative uncertainty (one definition of an interval) decreased while absolute uncertainty (another definition of an interval) increased as forest units were summed under an assumption of independence. This was because both median and variance terms increased linearly making the 10th to 90th percentile interval (which increased in proportion to the square root of the variance) an increasingly smaller proportion of the median.

Models structured to serve as accounting systems (for example, total forest carbon inventory) can be naturally organized into two sequential steps. First, determine a per-unit value of the quantity (for example, carbon per pool per hectare), and second, sum these units across an appropriate index (for example, forest area). This pattern appears in models (Nilsson and Schopfhauser 1995; Heath et al. 1996) and national summaries (Birdsey and Heath 1995; Kurz et al. 1995) as well as IPCC recommendations for greenhouse gas inventories (IPCC/OECD/IEA 1997b). Choices and assumptions made in the course of modeling affect the form and relatedness of intermediate PDFs, and these can affect final results.

Recommendations for pooling uncertainties often contain implicit but not clearly stated assumptions of independence (for example, Volume 1, p. A1.5, IPCC/OECD/IEA 1997b). Such relationships among uncertainties may be reasonable and accurate but could easily and inadvertently be hidden in assumptions as models are iteratively analyzed and revised. Clearly, issues of uncertainty continue to change and are unlikely to be entirely resolved–the state of science and the questions society asks of science change continuously. Therefore, a model structure that clearly and as simply as possible states basic assumptions is essential for subsequent use of uncertainty.

Decisions are seldom made on the basis of a single uncertainty analysis; generally, multiple influences need to be considered and merged by decisionmakers (Joint Climate Project 1992; Reckhow 1994; Klabbers et al. 1996; Paoli and Bass 1997). Probabilistic expressions resulting from analyses are useful to decisionmakers for considering multiple influences (Hoffman and Hammonds 1994; Morgan and Dowlatabadi 1996). A systems perspective is even more important when the array of external influences, and accompanying uncertainties, are considered. Global change will affect forest composition and growth as well as management practices and timber markets. Climate sensitivity and forest sector projections contain additional uncertainties that we plan to incorporate in our analyses. Where and how these added uncertainties appropriately link with the existing model can strongly influence rate of propagation.

Summary

Probabilities are commonly employed to quantify uncertainty. Discussion in this chapter focused on how summaries of probability distributions, or their subsequent use, can affect the interpretation of uncertainty. Model results presented here represent preliminary estimates of a portion of the uncertainty in carbon budgets for private U.S. timberlands.

Tractable use of results from uncertainty analyses often require tabular summaries of probability density functions (PDFs). The utility of a simpler format for expressing uncertainty should exceed the likely loss of information from a continuous distribution. Obviously, such summaries should still reflect the essential information desired by users. In other words, summarizing is fine, and understanding the form of the summary can help assure a net benefit. The relatively brief set of results we present here illustrate some basic considerations for ensuring this link and are summarized as follows:

- Tabular summaries (for example, "±10 percent") do not fully define distributions resulting from probabilistic simulations. Thus, summaries should focus on specific aspects of PDFs.
- Absolute and relative levels of uncertainty are useful summaries, yet they are distinctly different measures. Comparisons among estimates of uncertainty can be confusing unless definitions are clearly stated.
- A specified range for uncertainty includes an implicit assumption of likelihood based on the PDF. This should be explicitly stated as a range and associated confidence, for example.
- The use of a number of PDFs sometimes requires including additional characteristics in the summary, especially when summing a total uncertainty from separately obtained estimates. Size disparity and covariability among parts then become important considerations.

Probabilistic models, such as the implementation of FORCARB referenced here, explicitly account for such characteristics of PDFs. These guidelines are applicable whenever uncertainty is described in terms of probabilities, including policy and management decision making. That is, the use of probabilistic definitions of uncertainty requires many of the same considerations whether modeling or using the results from modeling. These are not complicated sets of rules but examples of the need for clear statements of definitions and assumptions.

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CHAPTER 8

Mitigation Activities in the Forest Sector to Reduce Emissions and Enhance Sinks of Greenhouse Gases

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International Negotiations to Stabilize Greenhouse Gases

In June 1992, representatives from 172 countries gathered at the "Earth Summit" in Rio de Janeiro to discuss environmental issues. The United Nations Framework Convention on Climate Change (FCCC) was adopted to achieve ". . . stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Such a level should be achieved within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure that food production is not threatened, and to enable economic development to proceed in a sustainable manner." The nonbinding goal of the Convention was "to return emissions of greenhouse gases to their 1990 levels by the end of the decade." The United States responded to the FCCC in 1993 with the "Climate Change Action Plan," a collection of about 40 individual programs covering emissions reductions, energy efficiency, and productivity enhancements including forestry activities.

At the first Conference of the Parties to the FCCC in 1995, it was concluded that voluntary commitments were inadequate and would not be met by most developed countries. Negotiators then agreed to the need for specific limits on greenhouse gas emissions beyond the year 2000. The U.S. position on mitigation of greenhouse gas concentrations was clearly stated at the second Conference of the Parties in 1996. Three elements were seen as necessary for ratification of a treaty: 1) realistic and binding targets; 2) flexibility in implementation; and 3) the participation of developing countries.

The third Conference of the Parties, held in Kyoto, Japan, in December 1997, produced an agreement known as the "Kyoto Protocol" that contained the first two elements: 1) binding targets, and 2) flexible implementation. The U.S. President promised to negotiate an amendment to the agreement covering the participation of developing countries prior to submitting the agreement to the Senate for ratification. Under the terms of the agreement, the United States is bound to reduce emissions of greenhouse gases 7 percent below 1990 levels by 2008-2012. This is a substantial reduction given that emissions are expected to rise substantially during this period due to population growth and economic expansion. Various countries and groups of countries have different reduction targets (and increases in some cases).

The role of forestry and land use change has been controversial throughout the international negotiation process. There are different opinions around the globe on whether forestry activities should be counted or not. A country's position depends on factors such as whether their forests are currently or prospectively a net source or sink for carbon dioxide (CO_2), whether carbon (C) stock changes in forests can be measured and verified, and the relative emphasis that should be placed on reducing emissions versus increasing sequestration. Some countries expressed concern that forest responses to "natural" factors such as increased atmospheric CO_2 (which may increase growth) would allow a country to claim credit for greenhouse gas reductions that are not associated with specific activities.

The Kyoto Protocol attempted to reconcile the diversity of viewpoints on land use change and forestry. According to article 3.3 of the Protocol, land-use change and forestry activities that can be counted toward the emissions reduction target include afforestation, reforestation, and deforestation since 1990 if the changes in stocks can be verified. According to most interpretations of the Kyoto Protocol, forest management activities alone are not sufficient to allow an area of forest to count toward the emissions reduction target. Article 3.4 provides an opportunity for nations to propose including additional activities such as forest management. The agreement does include sustainable forest management as part of a general statement supporting sustainable development and protection and enhancement of sinks.

The language, terminology, and accounting methods contained in the agreement are somewhat vague, and can be interpreted in different ways. Definitions of key terms such as "reforestation" are not stated, which becomes a problem for implementation because there are many different definitions in use throughout the world. The proposed accounting system is vague. For example, it is not clear whether harvested timber should be counted as a forest sink and if so, under which circumstances it could be counted.

To address these issues, the FCCC asked the Intergovernmental Panel on Climate Change (IPCC) to establish an expert panel to develop a special report on the land use change and forestry provisions of the Kyoto Protocol. That group reviewed definitions, accounting issues, and activities that could potentially be included within the terms of the Protocol, and documented the various options for eventual reconciliation during the ongoing Conferences of Parties.

This chapter addresses options in the United States forestry sector to reduce emissions of greenhouse gases and to increase the rate of carbon sequestration in forest ecosystems. We summarize the various options that have been proposed in the literature, review the methodologies used to analyze options and compute baseline estimates, evaluate the potential for implementing various options and the expected changes in emissions or sequestration, and review costs and other considerations in implementing mitigation policies.

Summary of Forestry Options to Reduce Emissions or Enhance Sinks

Numerous forestry options to mitigate atmospheric buildup of CO_2 have been proposed. These options are categorized below according to whether their primary or direct effect is on emissions reduction, sink enhancement, or a combination of emissions reduction and sink enhancement. Each of the options has indirect effects so that the three categories are not mutually exclusive. For example, forest management activities not only affect C storage in forest ecosystems, but affect the kind of products that may be produced from harvested wood, which in turn impacts energy use in two ways: 1) burning of byproducts to substitute for fossil fuel, and 2) substitution of wood products for similar products that use different amounts of energy in the production process (Marland et al. 1997).

Emissions Reduction

Reducing emissions is the most direct way to stabilize greenhouse gas concentrations in the atmosphere. Activities involving trees and forests may also achieve emission reductions indirectly, for example, by substituting one product for another, or by reducing demand for energy. In this section we identify the various forestry options for reducing emissions and the logic behind their potential inclusion as part of a comprehensive accounting for greenhouse gas sources and sinks.

Substitute Wood Products for More Energy-Intensive Products

Some wood products used in construction can be manufactured with less energy than non-wood substitutes such as aluminum and concrete (Skog et al. 1996). To the extent that such substitution is practical and economic, an increase in these wood products and a corresponding decrease in their substitutes reduces energy demand and associated emissions. The effectiveness of product substitution is based on a number of factors such as relative costs of inputs and elasticity of demand.

Reduce Demand for Energy in Growing Timber, Harvesting, and Wood Processing

Energy is used in establishing plantations, managing forests, harvesting timber, and manufacturing wood products. Efficiency of energy use can be increased through engineering at each step in the manufacturing process. Adoption of more energy-efficient practices depends on economic evaluation (U.S. Congress, Office of Technology Assessment 1991).

Reduce Biomass Burning (Wildfires)

Protecting forests from wildfire maintains standing biomass or allows biomass to increase. In some cases, particularly in the Western United States, fire protection has resulted in overstocked stands and large amounts of biomass in dead and dying trees, posing a substantial risk of catastrophic wildfire or other natural disturbance such as an insect or disease outbreak (Sampson and Clark 1996). Both the long- and short-term consequences of fire protection must be considered in evaluating this option.

Sink Enhancement

Sink enhancement technologies are designed to offset emissions by storing more C in forest ecosystems and wood products. Because much of the forest area in the United States is managed for timber products on recurring cycles of harvest, regeneration, and growth, there are opportunities to increase the average amount of standing biomass while still producing wood products. The harvested C that ends up in wood products and landfills is usually counted as an addition to the total amount of C sequestered. During the manufacturing process, wood waste that is burned for energy is sometimes counted to the extent that wood fuel is substituted for fossil fuel.

Afforest Marginal Cropland and Pasture

Conversion of cropland and pasture to forest, either by tree planting or natural afforestation, usually increases the amount of C stored in biomass and soils relative to the previous land use (Sampson and Hair 1992). If the new forestland is managed for wood products, then the disposition of C in wood products, byproducts, and landfills must also be considered.

Reduce Conversion of Forestland to Nonforest Use (Reduce Deforestation)

Conversion of forestland to nonforest use usually means loss of all or a substantial part of live biomass and reduction of organic matter in soils and the forest floor (Houghton 1996). CO_2 and other greenhouse gases are emitted when the removed biomass and organic matter are burned or decomposed. Some C may be sequestered for a time in wood products if the removed biomass is utilized. When part of a mitigation strategy, controlling deforestation is sometimes referred to as protecting/conserving existing forests (Matthews et al. 1996).

Improve Forest Management

There are opportunities to improve C storage by changing silvicultural practices on certain sites and forest conditions (Sampson and Hair 1996). The magnitude of increased C storage may be difficult to quantify since silvicultural practices are usually developed and applied for another purpose such as increasing timber growth and will not necessarily increase biomass growth. Nevertheless, some forest stands may not be growing at biologically potential rates because of severe overstocking or understocking, and these stands offer the best opportunities for enhanced C storage. Also, silvicultural practices may be designed to maximize the amount of C eventually stored in harvested wood products.

Reduce Harvest

The effectiveness of reducing harvest depends on temporal and spatial considerations. Reducing harvest can cause a short-term increase in the amount of C stored in forests because losses of C to the atmosphere during the removal of biomass and wood processing are avoided (Heath and Birdsey 1993). In contrast, over the long term, a continuous cycle of harvest, efficient utilization of biomass, and regrowth can sequester more C than not harvesting (Sampson and Hair 1996). The analysis should also address imports and exports between regions and countries since reduced harvest in one region may be offset by increased harvest elsewhere (increased imports) or by changes in wood processing technology.

Increase Agroforestry

Agroforestry can add biomass to otherwise low-biomass agro-ecosystems. It can also reduce the need to clear forestland for agriculture (Schroeder 1993). These C benefits can accrue along with increases in crop yields.

Combined Emissions Reduction and Sink Enhancement

Some technologies have potential to both reduce CO_2 emissions directly and enhance C sinks. Both effects must be analyzed to evaluate the potential contribution to greenhouse gas reduction.

Substitute Renewable Biomass for Fossil Fuel Energy

Short-rotation woody biomass crops may be grown specifically for energy reduction. When biomass is grown sustainably and used to displace fossil fuels, net C emissions are avoided since the CO_2 released in converting the biomass to energy is sequestered in the regrowing biomass through photosynthesis (Rinebolt 1996). Biofuels may be substituted for fossil fuels especially in the pulp and paper industry, which has access to waste biomass produced during manufacturing. There is not a one-to-one substitution because of differential conversion efficiencies and unpredictable energy markets.

Increase Proportion and Retention of C in Durable Wood Products

After harvest, forest C passes through a series of conversion processes to yield wood products and byproducts (Row and Phelps 1996). Maximizing the amount of C in products through efficient utilization of raw material, increasing the use of byproducts for energy substitution, and ensuring that unused byproducts are disposed in sealed landfills will minimize the amount of CO_2 emitted (see Skog and Nicholson this volume). Increasing the life of products in use may result in less new timber harvested for replacement products, which would affect C storage in biomass.

Increase Paper and Wood Recycling

Recycling wood fiber and wood products may reduce CO_2 emissions in two ways: 1) by reducing the area harvested to provide virgin fiber, and 2) by using less energy to convert recycled products versus growing, harvesting, and processing virgin fiber (Skog et al. 1996). Paper recycling is already common. Most solid wood products are currently disposed of in landfills and debris dumps and not recycled. Model estimates are used to quantify effects of recycling.

Plant Trees in Urban and Suburban Areas

Trees affect urban climate by shading, reducing wind, and evapotranspiration (McPherson and Rowntree 1993; Nowak 1993). Proper placement of trees and use of the correct tree species reduces the energy needed to heat and cool residential and small commercial buildings, with the magnitude of the energy reduction dependent on the local climate.

The U.S. Climate Change Action Plan

The U.S. Climate Change Action Plan (CCAP—Clinton and Gore 1993) was unveiled in October 1993 following several conferences to suggest and evaluate options. The plan's objective was to reduce greenhouse gases to 1990 levels by the year 2000 using cost-effective domestic actions. The plan consisted of nearly 50 individual actions affecting all significant greenhouse gases and all sectors of the economy. The plan was to be implemented voluntarily with $1.9 billion in new and redirected funding. Although the plan has failed to meet its goal because of strong economic growth, low energy prices, and funding shortfalls, the individual actions proposed in the plan were tried and evaluated, and the plan provides a basis for continuing efforts that are likely to become more important as the greenhouse gas problem worsens.

The plan included two domestic forestry actions to increase sinks (Moulton 1996). "Reduce the depletion of nonindustrial private forests" targeted poorly managed forests to ensure regeneration after harvest and maintain adequate stocking through landowner assistance programs. Cost was estimated at $4 million through 2000 for an expected emissions offset of 4.0 Tg C. "Accelerate tree planting in nonindustrial forests" was designed to increase tree planting by 233 thousand acres per year over the historical average of 2.5 million acres per year. This action was administered under the Forest Service Stewardship Program and was expected to cost $71 million through 2000. The amount of C sequestered by 2000 was expected to be a modest 0.5 Tg. The short time horizon makes tree planting appear to cost much more per Tg than reducing the depletion of forests. The amount sequestered from tree planting will increase substantially after 2000 because newly planted trees do not sequester C at a high rate until they are well established and have reached a fast growth stage.

The plan also included two domestic forestry actions to both increase sinks and directly reduce emissions. "Accelerate source reduction, pollution prevention, and recycling" included increased paper recycling, which both protects forest C by reducing harvest and reduces emissions because less energy is required to use recycled fiber versus virgin fiber. Including the non-forestry components, this action item was expected to cost $86 million through 2000 and reduce greenhouse gas emissions by 5.0 Tg C. "Expand cool communities program in cities and federal facilities" is based on strategic tree planting and lightening surfaces on buildings to reduce air conditioning energy use. The "Cool Communities" pilot program founded by EPA and American Forests would be expanded to 250 cities and communities and to 100 Department of Defense bases and other federal facilities. This activity was expected to cost $12 million through 2000, reduce greenhouse gas emissions by 4.4 Tg C, and sequester 0.5 Tg C in trees.

These four forestry actions continue to be part of the U.S. plan as described in the recent "Climate Action Report" (U.S. Department of State 1997). None of the actions achieved their original goal because the required funding was never made available.

An important international component of the CCAP is "Joint Implementation." Joint implementation allows U.S. and foreign partners to collaborate in meeting their obligation to reduce greenhouse gas emissions or increase sinks. These collaborative projects can sometimes achieve reductions more cost-effectively than if each country acted alone. For example, it may be less expensive to plant trees in a developing country, and the trees may grow faster than in some parts of the United States. There are many additional benefits to joint implementation such as sharing of technology, encouraging private sector development, and methodology evaluation.

Another international component of the CCAP is the U.S. Country Studies Program. This program is designed to: 1) enhance the ability of countries and regions to inventory emissions and sinks, and evaluate mitigation and adaptation responses; 2) enable countries to develop, implement, and monitor policies and measures; and 3) share information (Dixon et al. 1996).

Participation in the CCAP has been voluntary, with level of participation related to government incentives delivered through funded programs. Other incentives such as consumer preference are just beginning to be a factor. The U.S. Department of Energy sponsors a program called "Voluntary Reporting of Greenhouse Gases" that is developing the methodology and technology to collect and process data on the accomplishments of participants. There were 142 reporters in 1996 representing over 900 individual projects. Most participants have been electric utilities, although 20 percent are non-utilities.

The CCAP represents a first step by the United States to implement greenhouse gas mitigation activities. Although the CCAP has not met its goals, its implementation demonstrates that it is feasible to implement a program of emissions reductions or offsets and establishes partnerships to facilitate voluntary participation by consumers, companies, and non-federal government agencies.

Methodology for Estimating Mitigation Potential

Generally, analyses of forestry mitigation options attempt to determine the magnitude of expected gains in C sequestration and emissions reduction. Options help determine whether proposed activities are both biologically feasible and socially acceptable. The analyses must include sufficient detail at the national level so that policies can be evaluated with respect to societal concerns such as long-term trends in forest resources, economics of supply and demand, impacts on traditional and nontraditional forest products, energy tradeoffs, and land use changes.

The approach most often used to evaluate the mitigation potential of forestry activities involves analytical models that estimate the net effects of biological and social responses to implementation of a policy or activity. The expected C gains are estimated as a relative difference from "business as usual" or "baseline" scenarios. Integrating the biological and social components is critical for determining that the net effect of an activity is "additive," that is, a true departure from the expected baseline not including the activity.

In many mitigation studies, the complexities of ecological systems are represented in a highly simplified way based on observed data from inventories and ecosystem studies and from productivity estimates from a variety of forest growth models. Ecological process models that address the carbon cycle at large spatial scales (see Joyce et al. this volume; Bachelet et al. this volume) have not yet been fully integrated into mitigation analyses because they are usually validated for potential or equilibrium vegetation rather than managed or disturbed forest ecosystems, the subject of most proposed mitigation activities.

The complexities of social systems may be represented in several ways. Econometric models reflect past behavior as documented in historical data (see Mills et al. this volume). Past programs that were designed to implement forestry policies are often included as "case studies." Economic behavior can also be modeled by explicit optimization processes in markets (see later discussion of FASOM model).

The accounting system is a critical part of evaluating the various options (see Heath and Smith this volume). The accounting system should be comprehensive and include both positive and negative impacts on C. A comprehensive accounting system will be representative of the true impact of an activity on the concentration of atmospheric CO_2, whereas a partial accounting system may give misleading results. Comprehensive accounting is always difficult because of the many interactions among activities that preclude simple one-to-one estimation of additivity or substitution. The term "leakage" is often used to describe the difference between the direct effect of an activity on expected C, and the direct plus indirect effects that may occur through interactions.

Defining the scope or domain of the analysis is critical for quantifying the potential for mitigation. The critical domains are temporal, geographical, and sectoral. Temporal scale is important because activities that make sense in the short term may not make sense in the long term. For example, a short-term strategy of reducing timber harvest will increase C in forests for a few years but decrease C in wood products over the longer term. Also, there is increasing (cumulative) probability of damage from pests or fire as forests age, such that an event or series of events could result in large releases of C. The geographic scope is critical to addressing leakage because activities in one area (or country) may provoke an opposite (or reinforcing) action in another area. For example, reducing timber harvest on NFS lands in the Pacific Northwest may increase timber harvest from other regions (Adams et al. 1996b; Martin and Darr 1997).

Selecting which economic sectors to include and how to analyze outcomes across sectors may be the most complicated problem for addressing leakage. For example, increasing the use of biomass for fuel does not necessarily produce an equivalent reduction in the use of fossil fuels because energy markets are complicated globally and not driven completely by supply and demand economics (see Skog and Nicholson this volume; U.S. Congress, Office of Technology Assessment 1991).

Estimating the gains and losses in C associated with various options is also complicated by lack of data. For example, the impacts of forest management on soil C are poorly understood except in a few specific cases (see Heath and Smith this volume).

Finally, the interactions among various activities should be considered in a policy package. Different options may conflict or produce unintended consequences. For example, harvesting more timber to increase C in wood products is inconsistent with reducing harvest to maintain higher levels of C in forests. Both of these activities would have consequences for the nation's timber supply.

FORCARB and Forest Sector Models

The FORCARB model has several purposes: to estimate past, current, and prospective C storage and changes in C storage in U.S. forests and forest products; to simulate alternative policy options for enhancing the role of forests and forest products as C sinks; and to estimate how environmental change might affect C storage in forests and forest products (Plantinga and Birdsey 1993; Birdsey et al. 1993; Heath et al. 1996). FORCARB is one of a cluster

Table 8.1—Comparison of projected area changes for private timberland in the United States, from the TAMM/ATLAS/AREA CHANGE (T/A/A) and FASOM models, 1990 and 2000 decades (thousand acres). The afforestation, reforestation, and deforestation rates are on an annual basis. The private timberland total is as of the end of the decade.

	Afforestation		Reforestation		Deforestation		Total private timberland	
Decade	T/A/A	FASOM	T/A/A	FASOM	T/A/A	FASOM	T/A/A	FASOM
1990	1441	1674	4825	8022	1960	780	347,100	352,467
2000	558	916	5643	5293	936	710	344,000	354,529

Source: The projections are from baseline runs of two models: the TAMM/ATLAS/AREA CHANGE set is from the 1993 RPA Assessment Update (Haynes et al. 1994) and the FASOM projection is from a December 1997 run.

of integrated models of the forest sector that has been enhanced to evaluate global change effects on forests and wood products and to evaluate mitigation and adaptation strategies (Adams and Haynes 1996; Joyce et al. 1997). This integrated modeling system is used to simulate the effects of environmental changes on productivity, forest type transitions, harvesting, natural disturbance, timber production, and C storage. The system includes socioeconomic models used to conduct national assessments required by the Resources Planning Act (RPA). The socioeconomic models provide estimates and projections of human activities such as land use change and timber harvest that have major impacts on the status of forest vegetation.

The FORCARB model has the strength of national-scale, multi-sectoral analysis with sufficient representation of ecosystems, regions, ownerships, and management intensities to enable detailed analysis of options within a national policy context. A limitation is lack of linkage with the energy sector, so that energy inputs and outputs cannot be directly considered. The temporal domain is limited by the current model configuration that simulates future inventories about 50 years into the future.

FASOM Model

The Forest and Agricultural Sector Optimization Model (FASOM) described by Mills et al. (this volume) has been applied to examine the private forest management, land use, and market implications of terrestrial C sequestration policies (Adams et al. 1996a). The FASOM model uses the same empirically based timber yields from the ATLAS model as does TAMM and other forest sector models to which FORCARB is linked. While the models are similar in other regards, one key difference when examining policy options is that the FASOM model can estimate optimal land use and forest management investment in the context of mitigation strategies. This complements the positivistic approach of the TAMM system of models. A comparison of current and projected land use changes between FASOM and the TAMM system is presented in table 8.1.

When examining mitigation strategies involving forestry, increasing the area of forests and enhancing the productivity of existing forests are typical options to increase sequestration of C in forests and forest products. Many past studies have examined policy impacts of changing land use between forestry and agriculture. These studies typically have either: 1) ignored spillovers between sectors, or 2) simply "added up" impacts across the two sectors, ignoring feedbacks or interactions through the markets for land. To examine forest C sequestration policies while considering intersectoral competition for land, FASOM has both land use and forest management investment as endogenous decisions (Alig et al. 1997).

FASOM lacks linkage with the energy sector, so that energy inputs and outputs cannot be directly considered. The temporal domain is limited by computer resources, available data and assumptions, and policy interest.

Examples of Special Studies

American Forests, a nonprofit institution, organized two extensive studies addressing forests, global change, and mitigation options: increasing the area and growth of forests (Sampson and Hair 1992) and forest management opportunities (Sampson and Hair 1996). These studies brought together experts in many disciplines to evaluate options and provide guidance to public and private landowners for implementing opportunities for mitigation through forestry activities.

The U.S. Environmental Protection Agency has sponsored a series of studies that compared different models of mitigation options for U.S. forest and agricultural land (e.g., U.S. Environmental Protection Agency 1995a, 1997). These studies addressed scenarios of tree planting on marginal crop and pasture land; conservation reserve and wetlands reserve programs; increased use of recycled paper; reduced harvest on National Forest land; increased use of biomass energy; modified agricultural tillage practices; and increased use of winter cover crops.

The U.S. Congress, Office of Technology Assessment (1991) examined a suite of technical and policy measures to reduce greenhouse gases and determined that emissions of CO_2 could be reduced to as much as 35 percent below 1987 levels. Forestry activities comprised 10 percent of the reductions and included tree planting, increasing productivity, urban forestry, and use of trees for biomass energy.

The World Resources Institute studied U.S. forestry strategies to slow global warming (Trexler 1991). A mix of practices similar to the Office of Technology Assessment study was recommended.

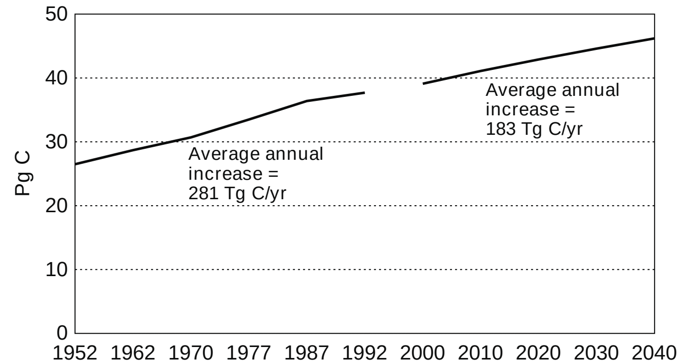

Figure 8.1—Past and prospective C storage for forests in the conterminous U.S. (from Birdsey and Heath 1995).

The Baseline Carbon Budget for U.S. Forestland

The "baseline" carbon budget refers to long-term trends in forest carbon storage using economic assumptions from the RPA Assessment (Haynes et al. 1995), in the absence of major forestry policy changes or changes in forest productivity or species distributions as a consequence of climate change. Long-term historical timber volume data converted to C estimates show that increases in biomass and organic matter on U.S. forestlands from 1952 to 1992 added 281 Tg C/yr of stored C to forest ecosystems, enough to offset 25 percent of U.S. emissions for the period (Birdsey and Heath 1995). Baseline projections using FORCARB show additional increases of approximately 183 Tg C per year in forest ecosystems through 2040 (fig. 8.1). The projected baseline includes forest policies in effect at the time the projections were made; in particular, reduced harvest levels on National Forest lands, decreases in clearcutting and increases in partial cutting practices, and continuation of federal cost-share programs at recent historical levels. Since that time, funding for cost-share programs was decreased.

The comprehensive baseline estimates are used as the forestry component of the "Inventory of Greenhouse Gases and Sinks" compiled annually by the U.S. Environmental Protection Agency (1995b). The EPA inventory includes forest C in living biomass, wood products, and landfills, and focuses on annual estimates beginning in 1990. The three forest components comprise an estimated annual sink of 125 Tg C for each of the years from 1990 through 1992. If C in the forest floor, coarse woody debris, and soils were added, the average annual estimate for 1990–1992 would be doubled to approximately 250 Tg C. These estimates do not include changes in Alaska, Hawaii, Puerto Rico, or U.S. Territories.

An earlier study converted 1987 forest area and volume statistics to carbon in standing biomass using simple models. Birdsey (1992a) concluded that U.S. forest trees were accumulating C at an annual rate of 461 Tg C, that removals from timber harvesting and land clearing totaled 355 Tg C, and that the annual net gain of C in live and standing dead trees totaled 106 Tg. Turner et al. (1995) used a similar approach but modeled some ecosystem components differently, particularly woody detritus. They estimated an annual accumulation of C in forest biomass of 331 Tg C, removals of 266 Tg C, and a net annual gain of 79 Tg C.

There are significant regional differences in past and projected C storage (fig. 8.2). These differences reflect variation in species composition and growth, as well as long-term changes in land use, management intensity, and harvesting practices. Millions of acres of forests in the Northeast have regrown on abandoned agricultural land, causing a steep historical increase in C, including a substantial buildup on C-depleted soils. As these regrowing forests mature, the rate of C buildup is expected to slow substantially. The historical pattern is similar in the South Central states, but the more intensive utilization of southern forests for wood products has already leveled past gains in C as growth and removals have come close to balancing. In the Pacific Coast states, C stocks are expected to increase after a recent decline, mainly due to reduced harvest projections as more forestland has been reserved from timber production.

The Kyoto Protocol (article 3.3) establishes a partial accounting system for forestry and land use change. The comprehensive forestry baseline would be changed to account only for forestlands that have been or will be affected by reforestation, afforestation, and deforestation since 1990. Forestry activities such as management and protection on lands not affected by one of these three activities would not be counted unless added under article 3.4. Since there is not yet agreement on interpreting the language, definitions, and accounting methodology of the Kyoto Protocol, it is impossible to calculate a new forestry baseline. The forestry baseline may change in several ways as illustrated in figure 8.3; however, the eventual baseline will likely be different from any of these as the interpretation of the Protocol evolves and partial accounting methods are implemented.

The alternative baselines in figure 8.3 are compared to the comprehensive baseline that accounts for all forestlands and all activities, as presented in Birdsey and Heath (1995). The first alternative accounts for the effects of reforestation, afforestation, and deforestation since 1990, with the important exception that the disposition of C in wood harvested prior to reforestation is ignored. Reforestation is defined broadly to include clearcut and partial cut harvesting followed by forest regeneration. The second alternative differs from alternative 1 by including the disposition of C in harvested wood. It is therefore a more complete accounting of the true impact of activities since 1990. Harvested wood that is burned for energy is counted as a source of C to the atmosphere and therefore deducted from the C sink estimate. The third alternative includes only afforestation and deforestation.

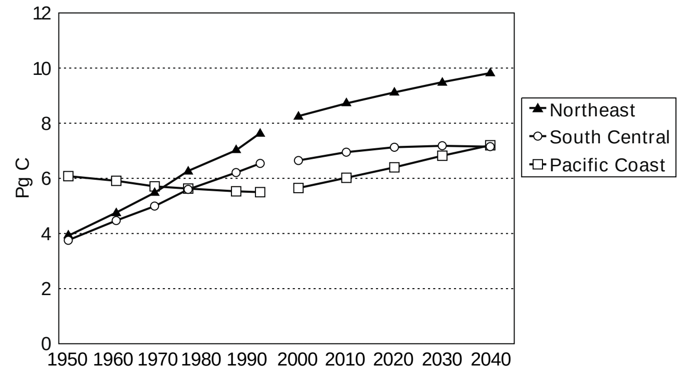

Figure 8.2—Past and prospective C storage for selected regions. Trends reflect land use history: maturing forests on reverted agricultural land in the Northeast; intensified timber utilization on reverted agricultural land in the South; reduced harvesting of old-growth and emergence of reforested areas in the Pacific Coast (from Birdsey and Heath 1995).

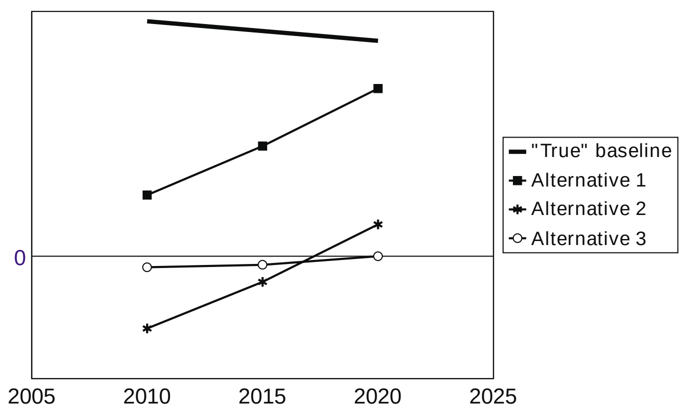

Figure 8.3—Illustrative simulation of several forest baselines from different interpretations of the Kyoto Protocol are compared to the "true" baseline that accounts for all forestlands and all activities. The first alternative accounts for the effects of reforestation (defined as broadly as possible), afforestation, and deforestation since 1990, with the important exception that the disposition of C in wood harvested prior to reforestation is ignored. The second alternative differs from alternative 1 by including the disposition of C in harvested wood and is therefore a more complete accounting of the effects of activities since 1990. Harvested wood that is burned for energy is counted as a source of C to the atmosphere and therefore deducted from the C sink estimate. The third alternative includes only afforestation and deforestation. The scale of the Y-axis is intentionally omitted.

Evaluation of Selected Mitigation Options

In this section we evaluate several mitigation options defined earlier as either sink enhancement or combined sink enhancement and emissions reduction. We do not evaluate options that are primarily intended to reduce emissions.

After a forest C baseline is established, the incremental effect of mitigation options can be evaluated relative to the baseline. The accounting system should include the effect of the activity on all C pools even if outside the forest sector. For example, C changes associated with deforestation should account for C retained in soils and biomass of the new land use. The studies reviewed here have not all used consistent ecological and economic assumptions and C accounting methods, and no attempt has been made to adjust reported estimates to a common basis. Nevertheless, the potential of some elements of a U.S. program to enhance forest C sinks are identified and their approximate costs established.

Afforest Marginal Cropland and Pasture

A large pool of non-forestland in the United States could be converted to forest to sequester additional C (Moulton and Richards 1990). Not all of the land that could support trees would be available, and the infrastructure may not be in place to provide seedlings for all available land. Large afforestation programs must be accompanied by increased nursery capacity. Additional technical assistance must also be provided to deliver

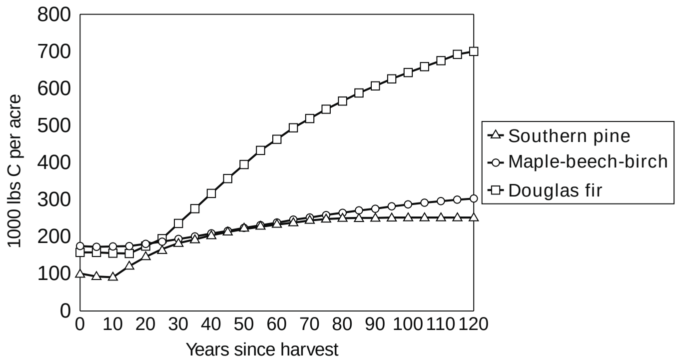

Figure 8.4—Comparison of C yields for several common forest types after a clearcut harvest: southern pine plantation on a good site in the South; maple-beech-birch forest in the Northeast; Douglas fir on a good site in the Pacific Coast (from Birdsey 1996).

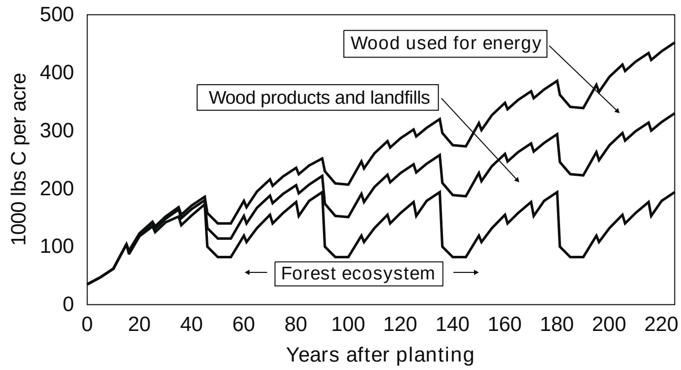

Figure 8.5—Pattern of C storage in a loblolly pine plantation managed for wood products over a long time period, including C in wood products and landfills (from Birdsey 1996).

planting programs effectively to landowners. Baseline projections by forest sector models already include substantial afforestation and reforestation amounts (table 8.1), so capacity and technical assistance issues would need to be addressed if additional afforestation efforts are directed specifically at forest C sequestration.

There is a time lag between tree planting and significant increases in C storage. Seedlings take several years to become established, and accumulation of biomass is low until trees reach sufficient size (leaf area) to fully utilize the "growth potential" of the site. As planted stands age, their growth rises, peaks, and then declines in a predictable pattern. The details of this pattern vary markedly by species, region, management regime, and potential catastrophic events such as fire, insects, and disease (fig. 8.4). For a one-shot afforestation program, aggregate C flux of the plantation would follow the pattern of the selected species. If timber stands are harvested for wood products and regenerated repeatedly over a long period of time, a sustainable pattern of increases and decreases of C in the forest becomes apparent (fig. 8.5). There is an accumulation of C in wood products and landfills over time as long as inputs to these pools exceed losses through decomposition. If wood used for energy is also counted, there is a further gain due to the substitution of wood energy for fossil fuel energy. Figure 8.5 illustrates the effect of a one-to-one substitution of wood energy for fossil fuel energy, an upper bound unlikely to be achieved when conversion efficiency and market effects are considered.

Many studies have estimated potential gains in C storage from afforestation. Moulton and Richards (1990) estimated that offsetting U.S. emissions by 10 percent (about 160 Tg C) would require about 71 million acres at an average cost of $12/ton of C or $1.7 billion/year. The U.S. Congress, Office of Technology Assessment (1991) estimated that a tree planting program on 3.5 million acres/year over 20 years would attain a net C flux increase of 30 Tg C/year by the end of the 20-year period. The annualized cost of this program would be about $35/ton C. Using estimates of C storage by age class for different forest types and conditions, Birdsey (1992b) estimated that converting 22 million acres of marginal cropland and pasture in the South to forest would eventually increase C accumulation by about 32 Tg C/year. Parks and Hardie (1995) estimated that converting 22 million acres of land to forest would increase C accumulation by 44 Tg C/year and cost $21/ton C.

These studies did not include effects of increased supply of timber on the forest sector, which may partially offset C gains by reducing prices and increasing quantity demanded. Parks and Hardie (1992) used FORCARB and forest sector models to develop two reforestation scenarios and compared the results with a base run (Heath and Birdsey 1993; U.S. Environmental Protection Agency 1995a). Planting was phased over a 10-year period from 1991–2000, and projections run through 2040. Most planting was expected in the South Central and North Central United States. The average annual increase in C flux (including C in wood products and landfills) over a 50-year period was projected to be 7.5 Tg C for a 0.7 million acres/year program costing $110 million/year, and 14.3 Tg C for a 1.2 million acres/year program costing $220 million/year. These are the direct costs associated with tree planting and payment of subsidies.

Projections using FASOM show that a 28 million-acre program (among other sector adjustments) costing an average $18 per ton C could produce an annual flux increase of 39 Tg C. Costs in this case are estimated as changes in social welfare. FASOM projections suggest that efforts to expand forest C flux should have a rather different geographic and species focus than that proposed in past studies. In contrast to both Moulton and Richards

(1990) and Parks and Hardie (1995), FASOM projections suggest a greater emphasis on hardwood species in minimum cost strategies. Hardwood area increases under all C targets (Adams et al. 1999). Some of this increase involves direct conversion of softwood to hardwood forests after harvest, but most derives from reductions in rates of hardwood-to-softwood conversion relative to the base case.

For some C policy scenarios, FASOM simulations indicate that the bulk of the projected afforestation and management changes should occur in the North, mostly in the Lake States region. This is an area of large concentrations of hardwood forests in which hardwood stands can yield significant rates of C uptake. Although the FASOM model recognizes the rapid growth potential of afforested stands in the South just as in previous studies, broader measures of costs and inclusion of welfare trade-offs across markets and regions act to partially shift the minimum cost solution away from the customary prescription of pine plantations on marginal Southern agricultural lands.

Opportunities for afforestation on nonindustrial private forestland are at least several times higher than recent historical rates. From 1994 to 1996, the U.S. private area planted annually to trees averaged about 2.28 million acres. As discussed earlier, there are tens of millions of acres where tree planting is biologically and financially feasible, especially on non-industrial private forestlands (Alig et al. 1990b; Vasievich and Alig 1996). In the FASOM projections a portion of those eligible acres are targeted for tree planting, particularly over the next two decades. For mitigation policy analysis, a key question is how many of the eligible acres are likely to be planted without any form of government assistance, and how much assistance would be required to induce additional plantings. If these opportunities were pursued, additions to forest C would be substantially higher than under the rates of afforestation projected in line with recent trends by the TAMM system (Haynes et al. 1994).

For large-scale afforestation programs, possible side effects include economic impacts from market dynamics (e.g., compensating land use changes from forestry to agriculture). Such effects can have significant influences on costs of C sequestration (Alig et al. 1997; Adams et al. 1999). Large-scale land use conversion could significantly alter opportunity costs in terms of foregone production from other land use alternatives (Alig et al. 1997). This may act to increase forest sequestration program costs and reduce C sequestration relative to that suggested in static or single sector studies (e.g., Moulton and Richards 1990; Parks and Hardie 1995).

FASOM simulations point to a somewhat different tree planting program than past experience indicates, suggesting more emphasis on hardwood species in the North and less emphasis on softwood species in the South. This finding based on a fuller accounting of opportunity costs highlights the need to carefully plan the implementation of any new C sequestration program by monitoring and re-evaluating economic conditions in the forestry and agricultural sectors.

Reduce Conversion of Forestland to Non-forest Use (Reduce Deforestation)

Approximately six million acres of non-federal forest in the United States (contiguous 48 states) were converted to urban and developed uses between 1982 and 1992 (U.S. Department of Agriculture, Natural Resources Conservation Service 1996). Another 6 million acres of forest were converted to agriculture and other uses. Further deforestation due to growth in urban and developed land is projected over the next several decades (table 8.1), as the United States is expected to add another 100 million people by 2050. Policy options for shifting land from agriculture to forestry for C sequestration must be viewed within the dynamics of land markets and historical trends in land use shifts. A combination of bio-physical, ecological, and socio-economic forces influence the amount of land allocated to major land uses and forest cover types in the United States. Population is the major factor influencing land use dynamics and the conversion of forestland to developed uses (Alig and Healy 1987).

Forest protection or conservation may also be included in this category of activities (Matthews et al. 1996). It may be difficult to determine whether a specific conservation project is truly a C offset activity if it is unclear whether the implementation of the project is due solely to a mitigation strategy, or would have occurred anyway (Brown 1998). Careful attention to identifying the factors included in the baseline calculation is needed to ensure that claims of C changes are truly relative to the baseline conditions.

Improve Forest Management

Timberland in the United States amounts to 490 million acres and includes a diversity of ownership objectives, forest types, site productivities, and stand conditions (Powell et al. 1994). There are opportunities to sequester additional C on some portions of this large area of forest. Of particular interest are opportunities to increase the density of trees on non-stocked or poorly stocked forestland, and to apply silvicultural treatments to stocked forestland so as to increase the average biomass per unit area. The changes in forest management intensity may be relatively small, but by affecting millions of acres of forestland, their aggregate effects may be large (Adams et al. 1999).

Many silvicultural practices are designed to increase the production of growing-stock volume in certain species. Gains in C storage are not necessarily proportional to gains in growing-stock volume because unmerchantable trees will also accumulate C, because stocking will increase naturally in poorly stocked stands, and because some management practices may remove biomass or disturb the site, resulting in loss of stored C. An analysis of broad management practices by major region and forest type in the United States concluded that strategies to maximize C accumulation should include: 1) converting poorly stocked forestland by clearing and regenerating only if current productivity is well below average; 2) applying intermediate stand treatments (thinning or timber stand improvement) only if the current stand is overstocked to the point of stagnation; and 3) managing for longer rotation lengths (Birdsey 1992c).

Including the value of C along with timber value changes the optimal economic rotation (Plantinga and Birdsey 1994; van Kooten et al. 1995). Both theoretically and in several case studies, the optimal rotation length increases if the benefits of C are counted. Harvest age was also found to change in FASOM projections in patterns that vary by species. For softwoods, rotations lengthen over all periods. Hardwood rotation changes are mixed and may, in some cases, involve reductions in both the near and long term.

Hair et al. (1996) summarized management opportunities for U.S. forests based on two comprehensive studies. They noted how timber and C yields varied significantly by management intensity. They concluded that managing plantations by means of timber harvest is the most effective way to achieve substantial and continual increases in C storage. Biological opportunities exist to increase timber growth (regeneration and stocking control) by 8.6 billion cubic feet on 202 million acres of timberland outside National Forests (Alig et al. 1990a; Vasievich and Alig 1996). Rates of return of 4 percent or more were available on almost half of these acres. Translating these potential gains in timber volume into gains in C storage is uncertain because of the variety of practices on many different species and sites, and because C gains are not proportional to timber volume gains. Nevertheless, Vasievich and Alig (1996) made a rough estimate that implementing the economic opportunities on timberland would yield gains in C storage of approximately 140 Tg C/year in vegetation, wood products, and offset fossil fuel C. Comparable gains from the biological opportunities were estimated as 190 Tg C/year.

Reforestation, defined as regeneration of forestland after harvest, may be natural or artificial (planted) in the United States. The definition of reforestation becomes synonymous with forest management for partial harvesting, a practice becoming more common in the United States since clearcutting has been reduced in the face of public opposition. Using U.S. Forest Service forest inventory statistics, W. Brad Smith (personal communication) estimates that between 1980 and 1990, 9.8 million acres/year were harvested, 62 percent by partial cutting methods. On National Forest lands, the area clearcut declined from 243,000 acres in 1984 to 133,000 acres in 1993. The area partially cut increased from 555,000 acres to 600,000 acres during the same period. At present, no studies have estimated how changes in harvesting and reforestation practices would influence C budgets at the national scale.

Conversion of mature or old-growth forest to young forest, which may have a faster growth rate, will reduce C storage until the harvested C remaining in products and landfills, plus additional C in the forest ecosystem from renewed growth, reaches the pre-harvest level. This may take 200 years or more in the case of old growth (Harmon et al. 1990).

Marland et al. (1997) analyzed the effects of forest management on C in forest ecosystems, wood products, energy substitution, and product substitution. Results of their model (GORCAM) suggest that over long time periods, sustainable management for forest products on highly productive sites will yield a larger C offset than simply protecting the forests intact. They note the difficulty of estimating the magnitude of the substitution effects, and of attributing the C offset to particular projects because the indirect effects of any given project are spread widely and are likely to be partly claimed as a credit elsewhere.

Reduce Harvest

Reducing the area harvested can cause an immediate short-term increase in the amount of C stored in forests because losses of C to the atmosphere during the removal of biomass and processing are avoided. On average, only about half of the live biomass is removed from the site, while logging debris (leaves, twigs, branches), stumps, roots, and unmerchantable biomass is left behind to several fates: decompose, transfer to another C pool (e.g., litter or soil), or become part of the new stand of trees (Birdsey 1992a). Of the biomass that is removed, about 35 percent ends up in durable products or landfills (based on removals since 1900 and historical patterns of utilization and disposal), while the remainder is burned for energy or emitted to the atmosphere (Heath et al. 1996; Skog and Nicholson 1998). Combining the estimates of on-site and off-site losses, less than 20 percent of the forest biomass ends up in long-term storage after harvest, and the remainder may be emitted to the atmosphere. Avoiding this loss by reducing harvest can be a short-term strategy to sequester additional C; however, over the long term, a continuous cycle of harvest, efficient utilization of biomass, and regrowth can sequester more C than not harvesting since the accumulation of C in the forest will eventually slow or stop, while it is possible to accumulate C in wood product and landfill pools for a very long time (Row 1996).

The effects of reduced harvest on C storage are evident in the estimated past and prospective C flux for National Forest lands (fig. 8.6, Birdsey and Heath 1995). High rates of harvesting in the 1970–1990 period caused emissions of 50 Tg C/year or more, while the significantly reduced harvest of the 1990s, if sustained, will cause a prolonged addition of C to National Forest lands, more than 80 Tg C/year. In the unlikely event that all harvesting were stopped in the United States, public and private timberlands could sequester an additional 328 Tg C/year over a 50-year projection (Heath et al. 1993).

Reduced harvest in one ownership category or region may be offset by increased harvest elsewhere, by substitution of energy-intensive non-wood products for wood products, or by changes in wood processing technology. Depending on the exact response, apparent gains in overall C storage may be lessened. The U.S. Environmental Protection Agency (1995a) concluded that reducing National Forest harvest by 21 percent would be fully offset by increased harvest from private timberlands and increased imports. Adams et al. (1996b) concluded that reduced harvest on public lands in the West could be largely offset by substantial private forest investment and increased harvest on private lands in the South. Martin and Darr (1997) found evidence for increased imports from Canada as a consequence of reduced National Forest harvest; but they also found inconclusive evidence for substitution of nonwood products or increased harvest on private lands.

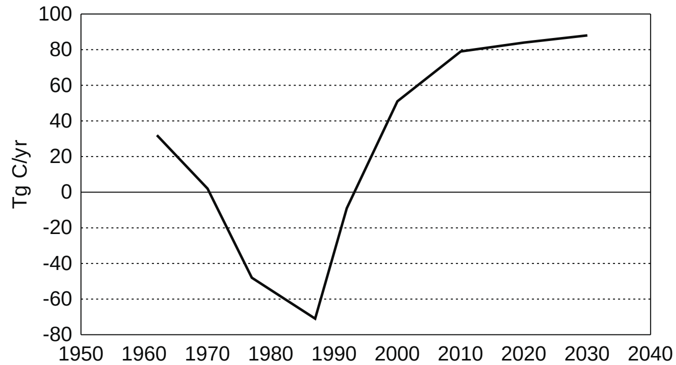

Figure 8.6—Past and prospective C flux on National Forest lands. Trends reflect high levels of harvest in the 1970s and 1980s, then a reduction in harvest in the 1990s resulting from legal and administrative requirements. Harvested C remaining in wood products and landfills is not included (from Birdsey and Heath 1995).

Substitute Renewable Biomass for Fossil Fuel Energy

Large quantities of wood are available for fuel from different sources: 1) residues or byproducts of wood product manufacturing; 2) roundwood not normally removed from timberlands during commercial harvest; 3) trees from "nonforest" areas such as fence rows and urban areas; and 4) roundwood (growing stock) customarily used for wood products (Rinebolt 1996). In addition to these existing sources, short-rotation woody crops could be established specifically for biomass production on marginal cropland and pasture (McCarl et al. in press). Current average dry biomass yields are approximately 5 tons/acre/year, with higher rates attainable (Wright and Hughes 1993).

The U.S. Congress, Office of Technology Assessment (1991) estimated that a program to plant about 1.25 million acres of biomass plantations per year for 20 years would eventually produce 30 Tg C/year of harvestable biomass. Estimating the potential C offset from use of this biomass is complicated by the uncertain availability of land, the relative conversion efficiencies of biomass and fossil fuel, and the actual displacement of fossil fuel by biomass. The OTA study estimated that about half of the harvested C would offset fossil fuel C. Wright and Hughes (1993) estimated that the conversion efficiencies of wood and coal to electricity are the same (33 percent), and that the net C offset averages 2.33 tons/ha/year for an average biomass production of 6.3 dry tons/ha/year.

Increase Proportion and Retention of C in Durable Wood Products

Knowledge of the disposition of harvested C is a critical component of evaluating forest carbon sequestration activities (fig. 8.5). The eventual disposition of wood and paper products in landfills should be included along with retention rates for products in use. Micales and Skog (1997) estimated that only 30 percent of the C from paper and almost none of the C from wood is ever emitted as landfill gas.

Heath et al. (1996) estimated that of the 10.7 Pg C harvested in the United States since 1900, 35 percent remained in products and landfills, 35 percent was burned for energy, and 30 percent was emitted to the atmosphere without producing energy for consumption. Heath et al. (1996) estimated that the current average net flux of C into products and landfills is about 37 Tg C/year, with 50 Tg C/year burned for energy or emitted. Skog and Nicholson (1998) estimated that, since 1910, 2.7 Pg C have accumulated and currently reside in wood, paper products, dumps, and landfills. Skog and Nicholson (1998) estimated that the 1990 rate of sequestration in wood and paper products, and dumps and landfills, was 61

Tg C/year. Harmon et al. (1996) estimated that of the 1.7 Pg C harvested from Oregon and Washington from 1900 to 1992, 23 percent is currently stored, primarily in structures and landfills. These estimates vary according to assumptions about historical patterns of harvest and product manufacturing, and disposal and retention rates in landfills and dumps.

Improved utilization of removed biomass could reduce losses of C to the atmosphere. For example, if the percentage of C in wood products were increased by 50 percent, the annual C storage in products would increase by about 10 Tg C, while the other disposition categories (landfills, wood burned for energy, and emissions) would each be reduced by about 3.5 Tg C/year (Heath et al. 1996).

Increase Paper and Wood Recycling

Increased recycling of wood products may have two effects: 1) keeping the C sequestered in usable products longer and 2) reducing the timber harvest. The U.S. EPA sponsored an analysis of recycling that concluded that each ton of recycled paper increased forest C sequestration by 0.73 tons (U.S. Environmental Protection Agency 1997). This estimate was derived from a cluster of U.S. Forest Service models including FORCARB and associated economic models of the pulp and paper industry. Another study estimated that rapidly increasing paper recycling to 45 percent of total fiber used would sequester an average of 10 Tg C/year (Heath and Birdsey 1993).

Plant Trees in Urban and Suburban Areas

Urban and suburban trees store C and can reduce energy use in buildings if the correct species are properly placed. Rowntree and Nowak (1991) estimated that urban areas in the United States have an average tree cover of 28 percent and store an average of 27 tons/ha. McPherson and Rowntree (1993) estimated that a single 25-foot tall tree can reduce annual heating and cooling costs of a typical residence by 8 to 12 percent, which both saves money and avoids the use of energy generated with fossil fuels.

Nowak (1993) concluded that planting an additional 100 million urban trees and maintaining them for 50 years would cumulatively store approximately 75 Tg C in biomass and offset 275 Tg C due to energy conservation. This is an annual average of 7 Tg C over the 50-year period. The rate of sequestration would be very low for the first two decades and higher toward the end of the period as the trees reach maturity (more than 10 Tg C/year). Assuming a cost of planting and initial tree maintenance of $5–25/tree (McPherson 1994), such a program would cost from $50 to $250 per ton of C after several decades.

Verification

Sequestered C may eventually have monetary value, be traded like other commodities, and be counted as an offset to C emissions in international treaties to limit greenhouse gas emissions. Therefore claims of C sequestration as a consequence of an activity must be accurate and verifiable. There must be internationally accepted ways to measure or estimate the gains and losses of C associated with specific activities. Estimates must reflect the true difference from a baseline that has resulted from a specific C sequestration activity.

Verification of attainment in increasing C storage requires an estimation and reporting system. The easiest way to estimate C gains at the national scale or for individual forestry projects is to measure the stocks of C at the beginning and end of a period of time. Unless expensive measuring equipment is used, 5–10 year periods are needed to measure changes in tree biomass. Soil C changes even more slowly, and both pool sizes and changes in pool sizes are more difficult to measure than tree biomass.

The net exchange of C between the ecosystem and the atmosphere can be measured over very short periods (minutes) using CO_2 flux measurement towers, but the equipment is expensive and the towers have been installed only under specific site conditions. Currently, estimates from a limited network of CO_2 flux towers are used to validate the regional and local estimates from forest inventories.

Birdsey (1996) estimated C storage by age class and ecosystem component for the major forest types in the conterminous United States, divided into nine regions. The estimates included the C stored in live trees, understory vegetation, litter and other organic matter on the forest floor, coarse woody debris, soil, and timber removed from the forest. The estimates cover 120 years beginning with the regeneration of clearcut timberland, cropland, or pasture. Carbon yield tables are reported for natural forest types and plantation species that are harvested and regenerated, and for pasture or cropland that is planted with trees or allowed to revert naturally to forest. Different site productivity classes and management intensities are included for some regions. All of the estimates represent expected regional averages for different vegetation classes (e.g., by forest type and past land use).

Carbon yield tables can be used to analyze the expected effects of specific activities outside the context of economic or policy models. The tables provide the basis for estimating changes in C storage in forests that would result from reforesting marginal crop and pasture land and increasing timber growth on timberland. The impacts of two of the action items in the President's plan for

Table 8.2—Estimated costs of forest carbon targets from various studies.

Study	Annual flux increase Tg C/yr	Land shift: agriculture to forests Million acres	Average cost: undiscounted carbon $/MT	Average cost: discounted carbon $/MT
Adams, et al. (1998)[1]	39	28	18	37
Moulton & Richards (1990)	23	9	9	—
	45*	21	10	—
Parks & Hardie (1995)	44*	22	12	—
	88	—	22	—
Richards, et al. (1993)[2]	44*	—	—	25
Adams, et al. (1993)	29	—	3	—
	56	50	7	—

[1] The forest carbon target scenario based on FASOM projections by Adams et al. (1999) involves a gradually rising carbon flux over a 100-year projection period, relative to the FASOM base case. The base case involves an increase in carbon flux of 1.25 gigatonnes per decade between the 1990 and 2000 decades, and a declining (but positive) rate thereafter. Other targets (not shown here) that require large near-term carbon flux increments have sharply higher costs than those that defer increases to later periods.

[2] Values estimated from figures for a 7.8 billion short ton program over 160 years. Costs vary with assumptions on discount rate, agricultural land demand elasticity, and agricultural land availability.

Source: This table is adapted from Adams et al. (1999). Scenarios with roughly equivalent average annual flux increment relative to base indicated by *.

reducing greenhouse emissions were estimated with C yield tables: 1) reducing the depletion of nonindustrial private forests and 2) accelerating tree planting in nonindustrial private forests (Clinton and Gore 1993). On the individual scale, guidelines for voluntary offsets proposed by the U.S. Department of Energy (1994) include tables similar to those that appear in Birdsey (1996).

Costs of Mitigation Policies

Recent national-level economic studies have examined the costs of attaining high rates of C storage to offset emissions (Moulton and Richards 1990; Adams et al.; 1993, Parks and Hardie 1995; Richards et al. 1993; Sedjo et al. 1995; Adams et al. 1999). In most of these studies, the sole vehicle for expanding C flux is the afforestation of agricultural land.

One of the earliest national-level studies that examined opportunities for mitigation activities in forestry was that by Moulton and Richards (1990) of the costs of reforestation and forest management for various levels of investment. They concluded that a maximum program level of $20 billion could offset about 56 percent of 1990 U.S. emissions (about 756 Tg C). The cost/ton of C would be about $10 for a 5 percent offset (67 Tg C) and about $18 for a 30 percent offset (405 Tg C). Cost estimates by Parks and Hardie (1995) are higher than those of Moulton and Richards (1990) in part because the former employ a smaller landbase. However, both studies do not consider interactions with existing forest inventories and markets; Parks and Hardie only consider afforestation options, while Moulton and Richards do include changes in management of existing forest.

Cost estimates with the FASOM model are generally higher than those from Moulton and Richards (1990) and Parks and Hardie (1995). Average costs per ton of C sequestered projected by the FASOM model are as large as twice those in the earlier studies (see table 8.2). This is due to rigid flux targets specified explicitly over time, recognition of intra- and intersectoral reactions to market changes, and inclusion of consumer impacts in welfare accounting (Adams et al. 1999). Costs are estimated as economic welfare losses in markets for forest and agricultural products. An example of the market-based considerations is the case of the C-target scenario projected with FASOM by Adams et al. (1999) that involves a gradually rising C flux over a 100-year projection period, relative to the FASOM base case. The base case involves an increase in C flux of 1,250 Tg C per decade between the 1990 and 2000 decades and declining (but positive) rates thereafter. Other targets that require large near-term C flux increments have sharply higher costs than those that defer increases to later periods.

FASOM-based findings of higher costs reflect, in part, the markedly different nature of the modeling approach. Earlier studies have generally focused on the process of shifting land from agriculture to forestry. The reckoning of costs has been limited to direct government payments to producers (for planting and rent subsidies) using a fixed schedule of agricultural land rental values. FASOM costs are net changes in surpluses in both agricultural and forest markets for consumers as well as landowners/producers, rent schedules are dynamic because of explicit product markets, and land may shift in both directions.

Another major departure from past studies is the inclusion of consumer-side impacts in FASOM cost accounting. Because of the linkage of the two sectors in the FASOM model, imposition of a flux target leads to countervailing land use and management responses in both sectors. From the cost perspective of earlier studies (that is, direct conversion and rent subsidy payments to agricultural land owners to afforest), recognition of these reactions could reduce the C gain for any given subsidy expenditure. For example, if afforested agricultural lands can ultimately be harvested, a land shift would raise agricultural land rents while lowering future forest products prices. This, in turn, would reduce both the incentive to maintain levels of forest management investment and to retain lands in forest cover rather than shifting them to agriculture (see Sedjo et al. 1995 for a similar discussion). Less intensive management or more forest-to-agriculture land movements would reduce the flux effects of the initial response. Ignoring these reactions, as in previous studies, would lower the apparent cost of the strategy.

FASOM cost results may also be higher than past studies because of the strict nature of the flux constraints. Previous work has focused mostly on afforestation or planting, accepting whatever flux time path that might result. While it is generally implied that policy "targets" are increases in average annual flux over some projection period, the length of this period is not always specified. And if the analysis allows harvest, the disposition of plantations after the first rotation is often not clear. The FASOM constraints eliminate this flexibility with attendant increases in costs. The FASOM projections do account for the storage of C in wood products after harvest, in contrast to the earlier studies. Storage in wood products can be substantial and warrants analysis of linked forest growth and harvest options.

Alternative approaches to estimating carbon sequestration costs determine how landowners actually respond to changes in net returns to forestry and agriculture (Plantinga 1997; Stavins 1996). Subregional studies (e.g., multi-county area) indicate that earlier studies may overestimate true costs of a carbon sequestration program due to failure to account for private non-market benefits from forests; however, costs may be underestimated due to failure to account adequately for option values and asymmetric information. Empirical results indicate that factors which tend to increase costs, such as option values, are more important than factors such as consideration of private non-market benefits that decrease program costs (Plantinga 1997).

Other Considerations in Policy Formulation

In addition to impacts on social costs, policy-induced land use changes may have other effects that should be considered in mitigation policy formulation. These include: 1) land use shifts to meet policy targets need not be permanent; 2) implementation of land use and timber management changes in a smooth or regular fashion over time may not be optimal; and 3) primary forms of adjustment to meet C policy targets involve shifting of land from agriculture to forest and more intensive forest management in combinations varying with the C policy target (Alig et al. 1997).

The benefits of sequestering C derive from elimination or reduction of potential damages resulting from future climate changes. Because there are likely to be lags between changes in C emissions, modifications in the climate, and effects on forests, it may be prudent, as part of a comprehensive review of policy options, to consider actions that entail large reductions in net emissions in the near term. In addition to the area drawn into the forest base through afforestation, obtaining these reductions could also involve changes in management practices on existing forests (such as rotation age) and altered intensities of management in future plantations on existing forestland or afforested areas.

Most previous studies have emphasized the physical changes and associated costs of forest C sequestration strategies. The studies have given little attention to the actual policy mechanisms or programs that might be required to implement the mix of actions indicated for a particular C flux target. This is a significant issue in that the costs or complexity of administering an otherwise ideal plan may preclude its use. Further, C cost estimates are frequently based on the normative assumption that landowners will accept the compensation for converting their land to forest (Plantinga 1997). Such compensation rates are assembled from a variety of data sources and often represent averages over broad geographical areas. The compensation rates do not account for some factors that may influence the decisions of landowners, including option values, private non-market benefits, and asymmetric information.

Analysis of forest C sequestration in the recent past has focused heavily on the impacts of expanded afforestation. Simulations of an array of specific intertemporal C sequestration targets using the FASOM model (Adams et al. 1999) suggest it may be cost-effective to supplement afforestation with other management changes. This is particularly so when policies require large increments in sequestered C in the near term. In these cases, rotation ages of existing softwood stands may be lengthened and new plantations employ a higher level of management input or intensity. Policies seeking more gradual increases in sequestration over the long-term, in contrast, rely more heavily on afforestation and a somewhat lower level of management input to these plantations.

A key long-term aspect of successful programs to shift land from agriculture to forest cover is the retention and condition of afforested areas. Empirical studies suggest that such afforestation plantations are retained at high rates over 10–15 years or longer, often exceeding 80 or 90 percent. These results have been consistent across the Soil Bank Program (Alig et al. 1980), the Agricultural Conservation Program (Kurtz et al. 1980), Forestry Incentives Program (Kurtz et al. 1996), and the Conservation Reserve Program.

Other considerations in policy formulation include infrastructural factors, degree of risk associated with forest investments, and relative difficulty in measuring C sequestration (Richards et al. 1997). An aspect of risk for C sequestration practices is timing of C uptake that results from a practice. For example, retaining a forest that is under imminent threat of clearing provides an immediate benefit—emissions that would have taken place in the near term are avoided. In contrast, the C uptake associated with afforestation can spread over several decades or even a century. If a government adopts a policy instrument that rewards the capture of C or avoidance of C release, the forest retention project will provide more immediate, and therefore less risky, returns (Richards et al. 1997).

Without careful analysis, C sequestration policies may have unintended negative effects. Implementation of forest policy instruments under real world considerations can sometimes lead to outcomes that differ significantly from those intended (Richards et al. 1997). One example from above is that basic market forces may be distorted by government intervention. Unforeseen links occur because we do not understand every possible outcome of a tax, subsidy, or other policy in advance. These types of market forces may in some cases offset, at least partially, land base and forest biomass changes intended by forest C sequestration policies (e.g., countervailing land transfers in response to concentrated large-scale afforestation programs).

Adaptations by humans is another consideration when designing mitigation policies. Policy deliberations should include how to facilitate adoption of appropriate forest production technologies and practices, including the cases where there may be beneficial effects of atmospheric CO_2 on tree growth. The forestry benefits of climate change are not likely to be equally distributed. For example, global warming in some areas, such as arctic and alpine areas, would likely increase the quantity of land suitable for forestry production. However, warming in other areas could reduce soil moisture, thereby shortening growing seasons and decreasing forest production.

Integrating C sequestration goals with those of broader forest policies involves emphasizing complementary benefits and examining values of C sequestration. Baseline projections indicate that U.S. forests and forest products will continue to add C storage (at a declining rate) through at least the year 2040. This baseline is based on optimization of a social welfare function, relying on market forces without any government intervention pertaining to C sequestration (e.g., C policy targets). In addition, integrating C sequestration goals with broader forest policies requires consideration of concerns over endangered species, biodiversity, and other forest-related services or goods. Policy analysts are not as well acquainted with and are less attentive to the unique considerations of forest C sequestration when formulating comprehensive policies. A current example of an opportunity for integrating policies is the Conservation Reserve Program (CRP) (Alig et al. 1997), which has been evolving into a policy with more environmentally-oriented objectives. Integrating C sequestration into the CRP objectives could result in significant afforestation of marginal pastureland and cropland and substantial C sequestration gains.

Conclusions: Potential for Mitigation through Forestry Actions in the U.S.

Forestry activities that directly or indirectly result in emissions reductions may play an important role in the ability of the United States to meet its international commitments to reduce greenhouse gases. The potential for increasing C storage in forests in the United States is quite large. Potential C storage is governed by the biological potential of forestland to maintain biomass, the availability of suitable land for forests, and the costs and tradeoffs associated with increasing and maintaining (protecting) a higher level of C in forests. Although it is practically impossible to maintain all forests at maximum growth and C storage simultaneously, there is a biological and economic potential to increase growth rates and the amount of C stored.

Projections indicate that even without a forest C program, substantial increases in forest C are likely consequences of current timber market activities and forestry policies. There is some uncertainty over time, especially if climate change impacts on ecosystems are substantial and cause catastrophic reductions in biomass as forest ecosystems attempt to adapt. Forest sinks are generally considered a short-term activity because of these limits.

Table 8.3—Summary of selected forestry options to increase carbon storage. Each option would be phased in over a 10-year period.

Option	Size of program	Change in C storage (Tg C/yr)	Annual cost (million $)	Years to achieve target
Afforestation of marginal cropland and pasture	23-45 million ac	50	350–770	20–30
Improve forest management	30-50 million ac	50	40–80	0–10
Reduce harvest	220 million cu ft	50	?	0–10
Increase recycling of fiber	from 40 to 45% of all fiber used	10	?	0–10
Increase C in durable wood products	Increase by 50%	10	?	0–10
Urban forestry	Plant 100 million trees	10	50–250	20–30
Increased use of biomass energy	1.25 million ac of plantations	30	?	10–20

But to the extent that reductions are needed sooner rather than later, forestry actions are an integral part of any comprehensive greenhouse gas reduction strategy.

Increasing the amount of C stored in wood products (in use or permanent disposal) is an important aspect of forestry activities. It is also possible to reduce greenhouse gas emissions from the forest sector by increasing energy efficiency in converting timber to products.

Size of programs, geographic location, and cost estimates vary widely because of differences in how past behavior is considered, differences in C accounting, and differences in model parameters. Carbon accounting rules will eventually become standardized. Models will continue to evolve, but since a model represents a particular view of possible future conditions, maintaining multiple models to allow for comparison of results from different perspectives will continue to be an important analytical activity.

Considering costs and potential impacts, and recognizing that some options have not been analyzed sufficiently, "improved forest management" appears to offer the most cost-effective means to sequester additional C in forest ecosystems in the short term (table 8.3). Verification of C changes attributable to forest management may be difficult because we lack sufficient experimental research that quantifies impacts of specific practices on different C pools.

Afforestation costs are high relative to reforestation, but considering the uncertainty of the estimation process and the fact that costs/ton increase as afforestation programs expand, some program level less than about 20 million acres could be cost-effective. Afforestation may also be needed to offset conversion of forestland to other uses (deforestation). The potential of afforestation is limited primarily by the availability of suitable land (for ecological or economic reasons), nursery capacity, willingness of landowners to participate, and availabilty of technical assistance.

Use of biomass energy will also be important, although we do not have good cost/benefit estimates available at this time. Some simulations have shown that biomass-fueled power is not very competitive with coal without subsidies. Substitution of wood products for other energy-intensive materials may also be effective, but estimating and attributing the benefits are difficult. Urban tree planting and energy efficiency in wood product manufacturing will both be important factors.

Protecting and conserving forests should maintain or increase C pools in the short term, as long as natural disturbance rates do not reach catastrophic levels. For any forestry activity, forest protection must be maintained or enhanced to sustain both the baseline rate of C sequestration and any investment in new programs.

Mitigation options can be analyzed most effectively within the context of the broad array of land use dynamics and forest cover-type changes that are driven by other factors besides forest C considerations. Possible unintended consequences of C sequestration policies warrant close attention by those formulating policies. Important considerations are possible effects on other sectors of the economy for large-scale and concentrated afforestation efforts, timing of C impacts from deforestation versus longer-term afforestation, and uncertainties in climate change projections.

Mitigation policies can not be evaluated independently of behavioral, economic, and institutional adjustments engendered by changing climate (Schimmelpfennig et al. 1996), both in the forestry and agriculture sectors. For example, if some agricultural producers respond to climate change by increasing the amount of land under cultivation, the amount of land available for forest C sequestration could be reduced. Within the forestry sector, producers may attempt to adapt to climate change by adopting appropriate tree planting mixes and practices. Further, increased research and technology transfer could promote technical advances that could help forest growers adjust to soil or other climatic characteristics. Long-run projections indicate that adaptations through forest C programs may not necessarily involve land use and forest management changes in a smooth or regular fashion over time, and that land use shifts to meet policy targets need not be permanent.

A number of policy tools involving forestry actions are available, including slowing deforestation to urban and developed uses and agriculture. Mitigation policies involving increases in forest C should be formulated with an awareness that a substantial increment to the U.S. population is projected to be added over the next several decades. Such population increases are likely to increase pressure to develop additional forestland (Alig and Healy 1987).

In this chapter, we have examined a range of mitigation options independently. Specific mixes of mitigation activities could be analyzed once more concrete policy targets are developed after the post-Kyoto deliberations move further along.

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