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# Forests and Carbon: A Synthesis of Science, Management, and Policy for Carbon Sequestration in Forests

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# Forests and Carbon

## A Synthesis of Science, Management, and Policy for Carbon Sequestration in Forests

Mary L. Tyrrell, Mark S. Ashton, Deborah Spalding,  
and Bradford Gentry, EDITORS



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# **Forests and Carbon**

**A Synthesis of Science, Management,  
and Policy for Carbon Sequestration in Forests**

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# Preface

The goal of this volume is to provide guidance for land managers and policymakers seeking to understand the complex science and policy of forest carbon as it relates to tangible problems of forest management and the more abstract problems of addressing drivers of deforestation and negotiating policy frameworks for reducing CO<sub>2</sub> emissions from forests. It is the culmination of three graduate seminars at the Yale School of Forestry & Environmental Studies focused on carbon sequestration in forest ecosystems and their role in addressing climate change. The seminars, part of the professional masters' degree curriculum, took place in 2008 and 2009. They were co-sponsored by the Yale Global Institute of Sustainable Forestry, the Center for Industrial Ecology at Yale, and the Center for Business and Environment at Yale. The seminars were led by Mark Ashton along with Bradford Gentry, Thomas Graedel, Xuhui Lee, Reid Lifset, Deborah Spalding, and Mary Tyrrell.

The purpose of the three seminars was to review and document what we know, what we do not know, and the implications for policy makers of: i) the science of carbon sequestration in forests; ii) the role of harvested wood products in the global carbon cycle; and iii) the science, business, and policy aspects of managing forests to store carbon. An overarching goal was to develop an understanding of the complexity of forest carbon science and why forest carbon budgeting has been a particular challenge for policy makers.

The basis of each seminar was a thorough review of the current literature on the topics, followed by in-depth class discussion. Leaders in the field were invited to give seminal talks, followed by lengthy discussion and debate with the class, to help set the stage for the students' review and analysis.

The resulting review papers, written by the graduate students under the direction of the faculty, are published in this volume. The collection provides a unique synthesis of current knowledge about science and management, and current thinking about policy, pertaining to the sequestration of carbon in forests globally. Overall, the volume supplies what we feel is much-needed scientific under-girding for discussions about carbon sequestration in forests. It contains recommendations for management and policy measures that reflect the scientific realities of how forests of many different types – tropical, temperate, and boreal – actually sequester carbon or do not, and under what circumstances.



We welcome comments and feedback – this is a work in progress amidst an evolving scientific understanding of a complex topic and an equally complex international dialogue on the role of forests in climate change mitigation. Please contact Mary Tyrrell ([mary.tyrrell@yale.edu](mailto:mary.tyrrell@yale.edu)) or Mark Ashton ([mark.ashton@yale.edu](mailto:mark.ashton@yale.edu)).

## Acknowledgements

The Yale School of Forestry & Environmental Studies (F&ES) funded three graduate seminars (2008-09) that were the source of materials in this volume, as well as the publication of this report. We are very grateful to former Dean Gus Speth, who enthusiastically supported the idea.

The Yale Global Institute of Sustainable Forestry has been a strong partner throughout, co-sponsoring the first seminar, *Forest Carbon Science*, and taking the lead in producing this volume. The Center for Industrial Ecology at Yale co-sponsored the second seminar, *The Role of Forest Products in the Global Carbon Cycle*, and the Center for Business and the Environment at Yale co-sponsored the third seminar, *Managing Forests for Carbon Sequestration: Science, Business, and Policy*.

We wish to acknowledge our colleagues on the F&ES faculty who co-led the seminars, hosted guest speakers, and reviewed many drafts of the papers. We appreciate their dedication to the success of the seminars and enjoyed the collaboration immensely.

Xuhui Lee, Professor of Meteorology

Thomas Graedel, Clifton R. Musser Professor of Industrial Ecology, Professor of Chemical Engineering, Professor of Geology and Geophysics, and Director of the Center for Industrial Ecology

Reid Lifset, Associate Research Scholar, Resident Fellow in Industrial Ecology, Associate Director of the Industrial Environmental Management Program, and Editor-in-Chief of the *Journal of Industrial Ecology*

F&ES faculty Graeme Berlyn, Ann Camp, Benjamin Cashore, Michael Dove, Timothy Gregoire, Florencia Montagnini, Chadwick Oliver, and Peter Raymond also reviewed several of the papers, which benefited greatly from their suggestions.

We are grateful for the assistance and suggestions of F&ES students Lauren Goers, who helped with editing and compiled the glossary, and Laura Bozzi, who was the teaching assistant for the spring '09 seminar. And thanks to Bertrand Tessa and Richard Campbell for GIS mapping assistance.

We benefited tremendously from our guest speakers, not only from their presentations, but especially from the discussions that followed. They enthusiastically engaged with the students, probing the depth of the issues. We thank those who took

time out of their busy schedules to travel to New Haven and all who shared their knowledge with the students.

Ralph Alig, USDA Forest Service

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And finally, we wish to acknowledge Jane Coppock, F&ES Publication Series Editor, and Dorothy Scott, who does page layout for the Series, for their hard work, dedication, and attention to detail, working under very tight deadlines, which helped make this project a success.

# Introduction

Lauren Goers,\* Mark Ashton, and Mary Tyrrell  
Yale School of Forestry & Environmental Studies

## WHY THIS REPORT?

The goal of this volume is to provide guidance for land managers and policymakers seeking to understand the complex science and policy of forest carbon as it relates to tangible problems of forest management and the more abstract problems of addressing drivers of deforestation and negotiating policy frameworks for reducing emissions from forests. It is an attempt at a comprehensive state-of-the-art review, encompassing the science of carbon sequestration in forests, management of forests for carbon and other values, and the socio-economic and policy implications and challenges of managing forests for carbon.

\* Yale Master of Forestry 2009

Forests are critical to mitigating the effects of global climate change because they are large storehouses of carbon and have the ability to continually absorb carbon dioxide from the atmosphere. But today, *emissions* from land use, land use change, and forestry, mostly due to deforestation in the tropics, are estimated at 17% of total annual global CO<sub>2</sub> emissions, a figure larger than the transportation sector (IPCC 2007).

While the basic principles of the carbon cycle are well known, there are significant uncertainties about the actual behavior of many of its sinks and sources. This is a particular challenge in forested ecosystems due to the role played by biogeochemistry, climate, disturbance, and land use, as well as the spatial and temporal heterogeneity of carbon sequestration across regions and forest types. The subject of forest carbon is complex, encompassing the science of carbon in forests, the economic drivers of deforestation, and the social and political contexts in which forests exist, making it a challenge to create comprehensive policies aimed at reducing CO<sub>2</sub> emissions from forests. Much work has been done on the science of forest carbon, deforestation, and various climate policy responses, including books, reports, symposia, and special journal issues (see for example, Streck et al., 2008; Griffiths and Jarvis, 2005; Angelson et al., 2008; IPCC 2000 Parker et al., 2008, among many others); however, what is lacking is a comprehensive review of all aspects of the challenge.

This book provides such a review by taking a holistic perspective on the subject. By creating a publication that outlines the research that has been done on forest carbon, pointing out what we know and what we don't know, and the implications for policy decisions, the hope is that land managers and policymakers alike will have a stronger foundation for making choices. The nature of the writing is meant to be accessible to a general audience and technical language has been simplified to the extent possible; nonetheless, this is a complicated topic with many “insider” terms. A glossary of scientific and technical terms is included at the end of the volume for quick reference.

## **Background**

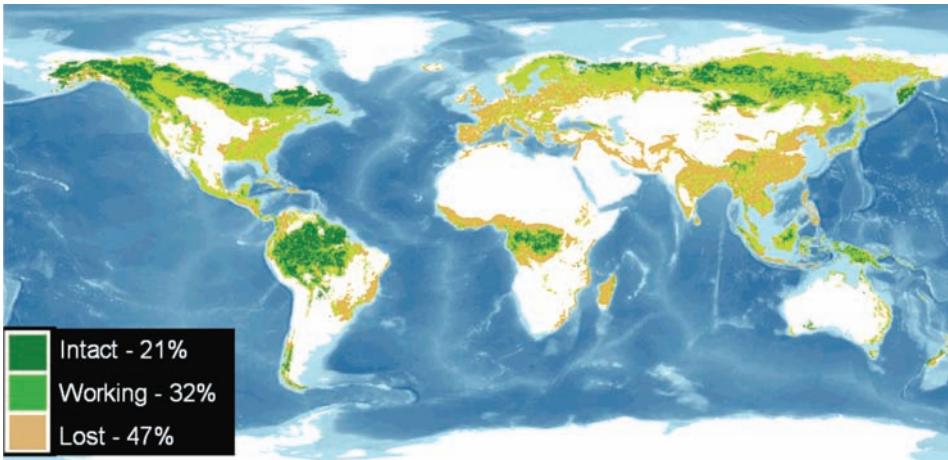
Forests are enormously important to maintaining global carbon sinks because they contain 77% of all terrestrial above ground carbon (IPCC, 2000; Houghton, 2007). At the United Nations Framework Convention on Climate Change's Conference of the Parties in December 2005, the governments of Papua New Guinea and Costa Rica introduced an agenda item on “reducing emissions from deforestation in developing countries and approaches to stimulate action” (UNFCCC, 2005). Since that introduction, the idea of addressing global increases in greenhouse gas emissions by reducing or avoiding tropical deforestation has been a topic that has sparked much debate in the international climate discussion. The need for a comprehensive strategy to reduce emissions from deforestation and degradation was subsequently reflected in the Bali Action Plan in December 2007 (UNFCCC, 2007). Since that time, discussion of how to implement a mechanism to “Reduce Emissions from Deforestation and Degradation” (REDD) has centered around questions regarding both the science of forest carbon and the design of sound policy informed by that science to achieve verifiable and lasting reductions in greenhouse gas emissions from forests.

Currently forests occupy just under four billion hectares of the Earth's land area, or roughly 30% of its land base. However, worldwide forest cover today is only a fraction of its historical extent, with some research estimating that 47% of original forest cover has been lost (Figure 1) (WRI, 2009). The extent of current net annual tropical deforestation is estimated at 7.3 million hectares each year (FAO, 2005). It is therefore imperative that forests be included in a global agreement to undertake actions for climate change mitigation and adaptation. Including forests as part of the global climate change mitigation strategy not only has climate benefits, but can help generate significant co-benefits, since keeping forests intact could also maintain biodiversity, preserve ecosystem services that many humans rely on, and help improve livelihoods of forest dwellers.

## **Contents**

The book is organized in three parts: the science of carbon sequestration in forests; management of forests and forest products for carbon storage; and the socio-economic, business, and policy aspects of managing forests for carbon.

**Figure 1** Historical forest loss and current extent of intact and working forests



Source: World Resources Institute (2009). Forest Landscapes Initiative. Reprinted with permission.

### *Science*

Part I focuses on forest carbon science. It examines carbon fluxes at varying spatial scales, from micro-sites to the global forest carbon budget, with particular attention on the impacts of such factors as climate, seasonality, disturbance patterns, and stand dynamics. It places this analysis within the context of broad forest types (boreal/temperate/tropical). It opens with Chapter 2, which analyzes research on carbon stocks and flows in forest soils, an important consideration for developing a forest carbon policy since two-thirds of the carbon in forests is in the soil (IPCC, 2000). Chapter 3 explores the underlying drivers of forest development and the ways these drivers are affected by changes in atmospheric carbon dioxide concentrations, temperature, precipitation, and nutrient levels. Chapters 4, 5, and 6 focus on carbon stocks and flows in boreal, temperate, and tropical forests, respectively, by reviewing both the literature on experimental research on carbon storage and flux in each biome, and models of predicted changes in regional climate, disturbance drivers, and effect on forest regeneration and dynamics in each forest type. Chapter 7 reviews methodologies for estimating carbon in above ground pools, a key topic for many nations in international policy discussions because of the need to develop standardized methods of carbon accounting with an emphasis on verifiable results. Part I closes with Chapter 8, analyzing the relationship between forests and the global carbon budget and describing current estimates and trends in the different stocks and fluxes of forest carbon.

### *Management*

Part II concerns the science and technology of managing forests for carbon sequestration and storage, including the life cycle of harvested wood products within managed forests and the advantages and disadvantages of accounting for wood

carbon stored or lost outside the forest. Chapters 9 and 10 describe the management and stand dynamics of forests for temperate and boreal, and tropical regions, respectively. Both of these chapters focus on assessing the impacts of silvicultural and management practices on carbon stocks and flows in various forest types. Chapter 11 focuses on the science of managing plantations and addresses key factors of implementing afforestation/reforestation projects for carbon sequestration such as site and species selection.

Chapters 12 and 13 assess the role of harvested wood products and the forest products industry within the context of global carbon stocks and flows, including life cycle analysis of forest products from harvest to end-of-life, and the implications for carbon storage.

### *Socio-economics and policy*

Part III concerns the socio-economic, business, and policy aspects of managing forests for carbon sequestration and storage. The first three chapters analyze the economic drivers of deforestation, focusing on deforestation for agriculture in the tropics (Chapter 14), threats to large intact forests (Chapter 15), and development pressures on forests in the United States (Chapter 16).

The final two chapters provide an overview of existing mechanisms and proposals for forest carbon policy at the global and U.S. federal levels. They describe the scale, reference levels, and financing for carbon projects in an attempt to broaden the understanding of current proposals and highlight key concerns for designing policy on forest carbon. Chapter 17 reviews both voluntary market mechanisms and forest carbon legislation in the United States and analyzes the scope, reference level, and proposed financing mechanisms for carbon offset projects. Chapter 18 looks at the forest carbon regimes proposed at the international level for inclusion in the climate treaty that is intended to replace the Kyoto Protocol in 2012.

## **CONCLUDING REMARKS**

At the end of each chapter and in the closing synthesis ideas, the authors have provided a summary of the most important conclusions from this review and their implications for forest carbon management or policy. These key points are designed to provide a guideline for developing strategies for managing forest carbon and developing a mechanism for reducing emissions from deforestation. The aim is to provide an accessible overview for resource professionals, such as land managers, to acquaint themselves with the established science and management practices that facilitate sequestration and allow for the storage of carbon in forests. The book has value for policymakers to better understand: i) carbon science and management of forests and wood products; ii) the underlying social mechanisms of deforestation; and iii) the policy options in order to formulate a cohesive strategy for implementing forest carbon projects and ultimately reducing emissions from the forest and forestry sector.

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# Part I: The Science of Forest Carbon

## SECTION SUMMARY

The following seven papers build upon each other to provide a comprehensive synthesis and review of the science of carbon in forests. The papers highlight areas of research that are well known and areas that are lacking. The first two papers cover soils and above-ground physiology and growth. They highlight the fact that most studies have been done in the temperate forests of developed nations.

The next three papers review the regional differences in carbon among tropical, temperate and boreal forest biomes. Studies show that tropical forests comprise nearly half of the total terrestrial gross primary productivity and that in recent decades Amazonian and Central African old growth forests continue to increase in biomass, which may be a response to increased atmospheric CO<sub>2</sub>. Temperate forests are mostly second growth and studies suggest that they are, on average, strong sinks for carbon, but a small change in temperature, rainfall or growing season length could change them from sink to source. The soil carbon pool plays a disproportionately large role in boreal forests, but increased fire frequency could greatly increase carbon release, with an even greater rate of heterotrophic respiration observed after fire.

The last two papers comprise an analysis of the different measurement techniques of carbon in the field and through remote estimation, and with this information global and regional statistics of stored and lost carbon are described. Four categories of methods for measuring forest biomass and estimating carbon are described: i) forest inventory (biomass); ii) remote sensing (relationship between biomass and land cover); iii) eddy covariance (direct measurement of CO<sub>2</sub> release and uptake); and iv) the inverse method (relationship among biomass, CO<sub>2</sub> flux, and CO<sub>2</sub> atmospheric transport).

Contributors toward organizing and editing this section were: *Mark S. Ashton, Mary L. Tyrrell, Deborah Spalding, and Xuhui Lee*

## Chapter 2

# Characterizing Organic Carbon Stocks and Flows in Forest Soils

*Samuel Price\* and Mark S. Ashton  
Yale School of Forestry & Environmental Studies*

### EXECUTIVE SUMMARY

Forests are expected to store additional carbon as part of the global initiative to offset the buildup of anthropogenic carbon dioxide (CO<sub>2</sub>) in the atmosphere (IPCC, 2007). Soil organic carbon (SOC) stored and cycled under forests is a significant portion of the global total carbon stock, but remains poorly understood due to its complexity in mechanisms of storage and inaccessibility at depth. This chapter first reviews our understanding of soil carbon inputs, losses from biotic respiration and the different soil carbon storage pools and mechanisms. Secondly, it evaluates methods of measurement and modeling of soil carbon. Thirdly, it summarizes the effects of diverse management histories and disturbance regimes that compound the difficulties in quantifying forest soil carbon pools and fluxes. Alterations of soil carbon cycling by land use change or disturbance may persist for decades or centuries, confounding results of short-term field studies. Such differences must be characterized and sequestration mechanisms elucidated to inform realistic climate change policy directed at carbon management in existing native forests, plantations, and agroforestry systems, as well as reforestation and afforestation. Such knowledge gains will also provide a theoretical basis for sound, stable investment in sequestration capacity. Lastly, the chapter provides recommendations for further research on those areas of soil carbon where knowledge is either scant or absent. Key findings of this review comprise what we do and do not know about soil carbon.

\* Yale Master of Forestry '08

### What we do know about soil carbon

Substantial work has been done that provides knowledge on many processes of soil carbon dynamics, such as:

- our understanding of how dissolved organic matter (DOM) additions from litter infiltrate the mineral soil.
- fine roots are the main source of carbon additions to soils, whether through root turnover or via exudates to associated mycorrhizal fungi and the rhizosphere.
- the dynamic between nitrogen deposition and carbon storage in forest soils is different on low-quality, high-lignin litter than on high-quality, low-lignin litter, which provides an explanation for many contradictory studies on the effects of nitrogen deposition.
- roots and mycorrhizal fungi produce about half of total respired CO<sub>2</sub>, with the balance from heterotrophic breakdown of organic matter.
- bacterial and fungal, as well as overall faunal community composition, have significant affects (+/-) on soil carbon dynamics.
- fossil fuel burning, particulate deposition from forest fires, and wind erosion of agricultural soils is expected to affect microbial breakdown of organic matter and alter forest nutrient cycling.
- organic matter can be stabilized from microbial action by biochemical resistance or by physical protection within soil aggregates or microsites – stabilization occurs from poor drainage (water logging), fire and deep charcoal burial, or stabilization of soil organic carbon (SOC), dependent upon the nature of the texture and mineralogy of the soil.

#### **What we do not know about soil carbon**

More research is needed to understand how the processes of soil carbon dynamics, that are now becoming understood, vary across different forest regions and soil depths. New research is needed that aims to characterize:

- controls on the depth of the organic layer by leaching of dissolved organic carbon (DOC) into the mineral soil.
- rates of fine root turnover among species and biomes.
- patterns of bacterial, fungal and plant respiration and responses to physical and biotic factors and stresses (such as drought, increased temperature).
- dynamics of functionally-distinct soil carbon pools, rather than the most-easily measured and fractionated pools.
- the most accurate methods for quantifying forest soil carbon stocks and fluxes.

The global nature of the carbon cycle requires a globally-distributed and coordinated research program, but thus far research has been largely limited to the developed world, the top 30 cm of the soil profile, temperate biomes, and agricultural

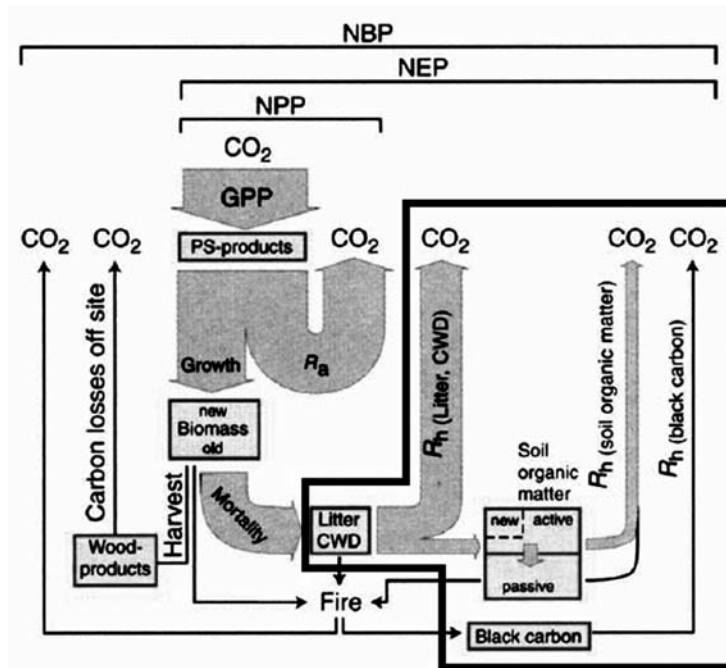
soils. Forest soils in tropical moist regions are represented by only a handful of studies and even fewer have examined sequestration of mineral carbon at depth.

**Keywords:** SOM, SOC, sequestration, respiration, climate change, mitigation, soil science, forestry, forest management

## INTRODUCTION

Carbon enters the terrestrial biosphere only through photosynthesis, and is shunted to the soil system by leaf- and debris-fall, the turnover (cycle of death and new growth) of roots, and by the allocation of plant photosynthate to mycorrhizal fungi. Plant residues are broken down by bacteria and saprophytic fungi, resulting in a cascade of complex organic carbon compounds that leach deeper into the soil. Carbon that leaves the forest soil system exits almost entirely via  $\text{CO}_2$  respired by plants, bacteria and fungi (Figure 1).

**Figure 1 Forest carbon flux.** The black box outlines the limits of the belowground carbon cycle. Arrows represent fluxes and boxes indicate pools; the size of each indicates the relative rate of flux or size of pool. Litter and coarse woody debris on the forest floor are included in the belowground portion of the forest carbon cycle. NBP = net biome productivity; NEP = net ecosystem productivity; NPP = net primary productivity; GPP = gross primary productivity; PS = photosynthesis;  $R_h$  = heterotrophic (bacterial) respiration;  $R_a$  = autotrophic (plant and associated mycorrhizal fungal) respiration; and CWD = coarse woody debris.



Source: Schulze, E.D., Wirth, C., Heimann, M., 2000. Managing forests after Kyoto. *Science* 289: 2058-2059. Reprinted with permission from AAAS.

These divergent respiration pathways differ in rate, substrate preference (e.g. type of litter, root or woody debris), and response to environmental change, complicating our capacity to characterize them. Carbon that remains in soil does so because it is stabilized by its own intrinsic chemical properties, by physical separation from microbial breakdown, by molecular interactions with metals or other bio-molecules, or by freezing, inundation from flooding or carbonization.

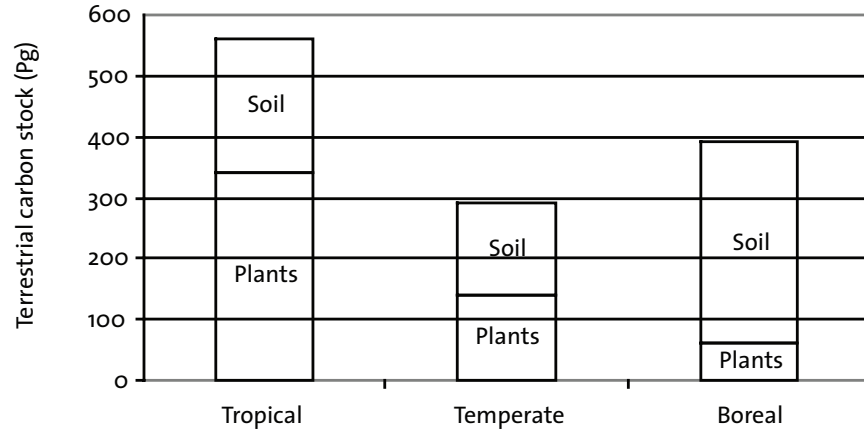
This is an introductory summary of the portion of the carbon cycle that is closely linked and affected by soil forming processes in terrestrial ecosystems, and most importantly, in forests. What is clear is that soil carbon, as a component of the ecosystem, varies enormously across different forest biomes (Figure 2), and across different soil orders (Figure 3).

**Figure 2 Distribution of world forest carbon stocks by biome.** Tropical forests worldwide contain approximately as much carbon in living plants (340 Pg) as boreal forests contain underground (338 Pg), indicating broad differences in carbon dynamics between biomes.

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In general, soil carbon is strongly associated with rainfall distribution and therefore there is more carbon stock in forests than in other terrestrial ecosystems

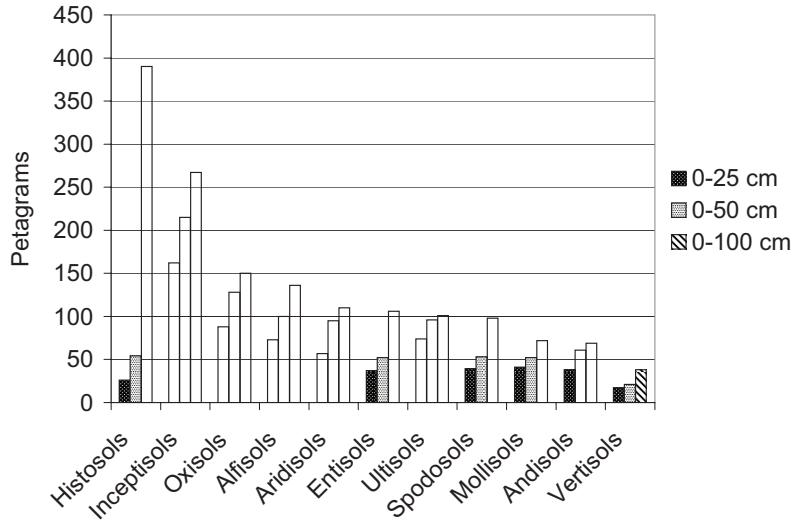
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Source: Data compiled from Vogt et al., 1998; Eswaran et al., 1995; Goodale et al., 2002; Guo and Gifford, 2002.

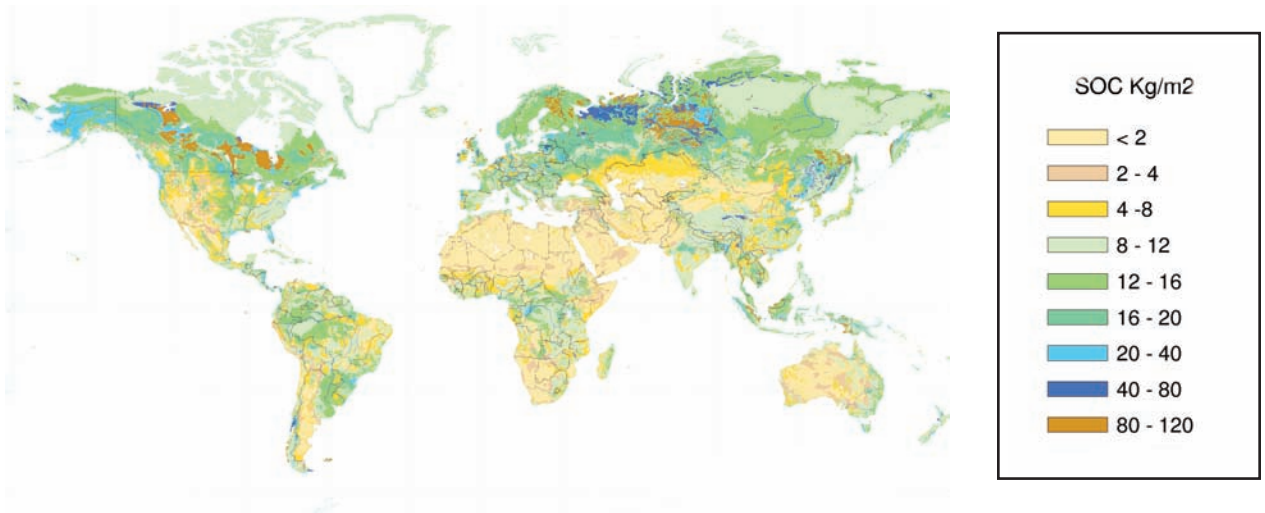
In general, soil carbon is strongly associated with rainfall distribution and therefore there is more carbon stock in forests than in other terrestrial ecosystems (Figure 4). The nature and condition of forests, by implication, can therefore play a critical role in soil carbon sequestration and storage processes. In this chapter the carbon in soils is described in the form of inputs, losses, and as that portion of carbon that remains stable within soil. It proceeds with a review of methods of quantifying soil carbon processes and pools directly with measurements and through modeling. It concludes with a discussion of effects of management on the carbon in forest soils and finally makes recommendations on what further research and knowledge is needed and where.

**Figure 3 Soil organic carbon (SOC) stocks worldwide, by soil order.** Histosols store the majority of the world's SOC due to seasonal or continuous inundation, and do so at depths between 50 and 100 cm.



Source: Adapted from Eswaran et al. (1995)

**Figure 4 Density of soil carbon stocks worldwide.** Note the swaths of highest density across the boreal regions of North America, Europe and Asia. Across the boreal forest SOC stocks are spatially variable.



Source: US Department of Agriculture, Natural Resources Conservation Service, Soil Survey Division. Washington D.C. <http://soils.usda.gov/use/worldsoils/mapindex/soc.html>. Reprinted with permission.

## **CARBON INPUTS TO FOREST SOILS**

Plants take in carbon dioxide and produce sugars under photosynthesis. Photosynthetic products are used to drive cellular respiration, storage for future consumption, reproduction, or allocation to root, shoot and wood growth. When leaves, branches or roots outlast their useful life and cease to provide a net contribution to plant growth, they die. Plants thus control the input of carbon to the soil system via above- and below-ground carbon inputs into forest soils from plant litter, coarse debris, fine root turnover, and root exudates.

### **Aboveground carbon inputs: litter and coarse woody debris**

Carbon from aboveground sources enters the soil system when it falls to the forest floor in the form of dead leaves, bark, and wood. Carbon is lost from surface organic matter as CO<sub>2</sub> by microbial respiration, by mixing and incorporation of surface organic matter into mineral soil horizons by soil fauna, and by leaching of dissolved organic matter (DOM) of which dissolved organic carbon (DOC) is an important constituent.

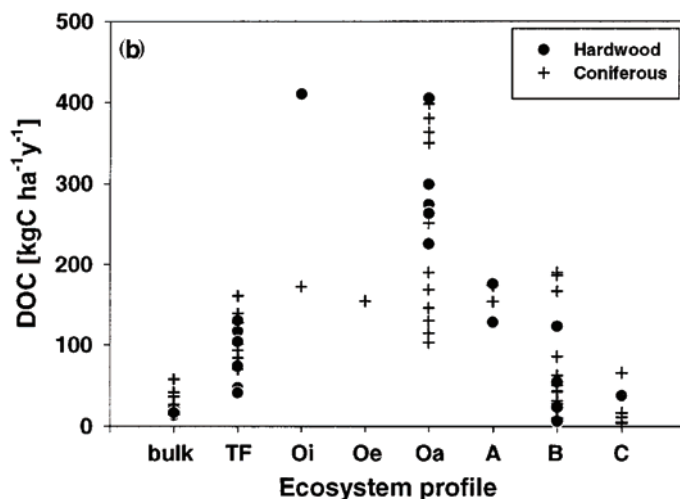
In a synthesis of 42 studies from temperate forests, Michalzik et al. (2001) reported that precipitation was strongly positively correlated with the flux rate of DOC from the forest floor into the mineral soil. The concentration of DOC in leachate from the forest floor to the mineral soil was positively correlated with pH, suggesting that more basic conditions favor microbial decomposition and thus DOC production. They also found that the greatest annual fluxes and greatest variability were in the lowest humified organic layer (Oa). There were very few studies of DOC flux from the upper organic layers. DOC flux decreases with depth in the mineral soil. There was a significant contribution of DOC from throughfall (TF), a result of microbial breakdown of organic matter in the canopy. There was no significant difference between DOC fluxes under coniferous versus deciduous forest (Figure 5). More recent <sup>14</sup>C labeling studies from Sweden and Tennessee, USA, corroborated these results (Froberg et al., 2007a; Froberg et al., 2007b), indicating that most litter-derived DOC is either respired before it reaches the mineral soil or immobilized in the Oe and Oa surface layers of the soil (Figure 5).

In a litter manipulation study at a hardwood forest in Bavaria, Germany the net loss of DOC from organic horizons was related to depth of those horizons rather than microbial respiration. DOM is continually leaching through the soil profile, such that leachate at any depth will be a combination of new litter-derived DOM and older DOM released from humic or lower layers (Park and Matzner, 2003). DOC from older litter showed a higher contribution of carbon from lignin and lower biodegradability relative to fresh litter (Don and Kalbitz, 2005; Kalbitz et al., 2006). Conflicting results from laboratory and field studies have been hard to reconcile because of lack of controls for hydrology as well as nitrogen and phosphorus (Kalbitz et al., 2000).

Different physical properties of litter affect microbial colonization rates and thus breakdown (Hyvonen and Agren, 2001). Litter, coarse woody debris, and roots of trees show differences in chemistry, rates of mass loss of litter due to decomposition,

and nitrogen dynamics by species. Recently, some researchers suggest that increased atmospheric CO<sub>2</sub> might lead to altered degradability of organic matter due to chemical changes in leaf or root chemistry (Hyvonen and Agren, 2001). But it appears from Free-Air CO<sub>2</sub> Enrichment (FACE) studies that species-specific differences in organic chemistry (e.g. pine versus birch) outweigh changes due to CO<sub>2</sub> enrichment. For temperate forests at least, changes in species competitive growth advantages due to heightened CO<sub>2</sub> will be the real driver of change to DOC dynamics (King et al., 2001; Finzi and Schlesinger, 2002; King et al., 2005; Hagedorn and Machwitz, 2007). Barring limiting nutrients or water, litterfall (leaf productivity and turnover) is expected to increase under heightened atmospheric CO<sub>2</sub> without a concomitant change in litter chemistry (Allen et al., 2000).

**Figure 5 Synthesis of 42 studies of DOC from the temperate forest biome showing annual fluxes of DOC through the organic and mineral soil profile.** The greatest annual fluxes and greatest variability are for the lowest humified organic layer (Oa – soil organic layer). The figure depicts a lack of studies of DOC flux from Oi and Oe layers. DOC flux decreases with depth in the mineral soil. Note the significant contribution of DOC from throughfall (TF), a result of microbial breakdown of organic matter in the canopy. There was no significant difference between DOC fluxes under coniferous versus deciduous forest. Bulk = bulk precipitation; TF = throughfall precipitation; Oi = litter layer; Oe = fermented layer; Oa = humic layer; A, B and C = successively deeper mineral soil horizons.



Source: From Michalzik, B., Kalbitz, K., Park, J.H., Solinger, S., Matzner, E., 2001. Fluxes and concentrations of dissolved organic carbon and nitrogen – A synthesis for temperate forests. *Biogeochemistry* 52, 173-205. Reprinted with permission.

Recent work suggests an interesting dynamic between nitrogen deposition and carbon storage in forest soils: Under nitrogen deposition on low-quality, high-lignin litter, decomposition of the organic layer slows, while nitrogen deposition on high-quality, low-lignin litter tends to accelerate decomposition (Knorr et al., 2005a). This dynamic provides an explanation for many contradictory studies on the effects of nitrogen deposition. A long-term study in Michigan, USA, demonstrated that chronic



nitrogen additions increase soil carbon storage through reduced mineralization of surface and soil organic matter (Pregitzer et al., 2008), although a contrasting study indicated increased litter mass loss in high-nitrogen microcosms (Manning et al., 2008).

### **Belowground carbon inputs: fine root turnover and exudates**

Fine roots are the main source of carbon additions to soils, whether through root turnover or via exudates to associated mycorrhizal fungi and the rhizosphere. Quantifying fine root turnover in-situ is therefore important but difficult because of their dynamic but variable turnover rates. Previous studies had indicated an extremely rapid turnover of fine roots, on the order of months to just a few years (Vogt et al., 1998). More recent studies using radiocarbon dating, however, indicated that roots were turning over on a 5-10 year cycle (Trumbore, 2006). These opposing observations can be reconciled if the distribution of root ages is assumed to be positively skewed, with many small and ephemeral roots turning over in a matter of weeks, with a long tail of older roots surviving upwards of two decades (Trumbore and Gaudinski, 2003). Results underline the need to conceptualize and model root turnover with multiple root pools rather than a single pool with a universally-applied turnover time (see below for a discussion of problems encountered in determining the rate of fine root turnover).

While DOM additions from litter have been extensively researched and reviewed, the fate of DOM additions from fine roots has been investigated by just one study to date (Uselman et al., 2007). Litter at the soil surface microcosm lost the most carbon during the study, with decreasing percentage loss with depth of litter addition, suggesting an important role for deep roots in adding stable carbon to the soil system (Uselman et al., 2007). A large scale tree girdling experiment in a Scots pine forest in Sweden resulted in a 40% drop in DOC, suggesting that current photosynthate contributes significantly to soil DOC through ectomycorrhizal fungi growing in association with roots (Giesler et al., 2007). This finding contrasts with the popular paradigm that DOC is primarily the product of root decomposition, since DOC should have increased following girdling had decomposition been the primary avenue for DOC production (Hogberg and Hogberg, 2002; Giesler et al., 2007). A recent Free Air Carbon Dioxide Enrichment (FACE) experiment documented fully 62% of carbon entering the SOM pool through mycorrhizae turnover (Godbold et al., 2006), which may explain the close link between recent photosynthesis and DOC additions to soil. Results thus far are ambiguous on the impact of elevated CO<sub>2</sub> on fine root production and turnover, with some studies indicating modest positive increases in root productivity (Luo et al., 2001b; Wan et al., 2004), while others show little or no increase (Pritchard et al., 2001; King et al., 2005; Pritchard et al., 2008). A recent study also showed that elevated atmospheric CO<sub>2</sub> does not cause changes in fine root chemistry, specifically increases in recalcitrant compounds that previous researchers had suggested may slow decomposition (King et al., 2005).

## CARBON LOSS THROUGH ROOT, FUNGAL, AND BACTERIAL RESPIRATION

Only in the last fifteen years, with our ability to accurately measure respiration of micro-organisms in field circumstances, has attention been given to understanding processes of carbon loss in soils. This section summarizes the more recent work done on root exudates, decomposition, and fungal and bacterial activities contributing to carbon loss from forest soils.

### Root, fungal, and bacterial respiration

Roots and mycorrhizal fungi produce about half of total respired CO<sub>2</sub>, with the balance from heterotrophic breakdown of organic matter (Ryan and Law, 2005). Soil respiration is commonly partitioned between autotrophic (plant) and heterotrophic (decomposition) respiration. These lumped categories simplify complex relationships in the soil system. For example, ectomycorrhizal fungi are clearly not primary producers, yet respiration products from ectomycorrhizal fungi are lumped with autotrophic respiration due to their close coupling with root processes and dependence on recent photosynthate. Respiration is more accurately viewed as a spectrum from fully autotrophic photosynthesizers to fully heterotrophic predators and decomposers (Ryan and Law, 2005). Conceptual models and new techniques for partitioning soil respiration among sources are needed (Table 1).

**Table 1** Experimental methods employed to date for partitioning soil respiration among autotrophic and heterotrophic sources

Category	Technique	
Root exclusion	Trenching	All roots crossing the perimeter of the treatment plot are severed; membrane installed to prevent regrowth
	Girdling	Girdled trees near or within treatment plots cannot allocate photosynthate to roots
	Gap	Compare soil CO <sub>2</sub> efflux in clearcut stand to control stand
Physical separation of components	Components	Separate litter, roots and mineral soil from a soil core; incubate separately; measure CO <sub>2</sub> efflux from each component
	Root excising	Remove roots from a fresh soil core; measure CO <sub>2</sub> efflux immediately
	Live root respiration	Excavate roots while still attached to tree; isolate and measure CO <sub>2</sub> efflux <i>in situ</i>
Isotopic techniques	Isotopic labelling	<sup>13</sup> C labeling in FACE or chambers; switch C <sub>3</sub> with C <sub>4</sub> plants
	Radiocarbon	Radioactive decay of <sup>14</sup> C permits dating of photosynthetic event
Indirect techniques	Modeling	Bottom-up simulation of response of soil components to biotic and abiotic factors
	Mass balance	Assume soil C is at steady state; Measure rates of C addition to soil from above- and belowground sources; subtract soil CO <sub>2</sub> efflux
	Subtraction	Soil CO <sub>2</sub> efflux minus other flux components from ecosystem NPP models and published values
	Root mass regression	Regress CO <sub>2</sub> efflux at multiple sites against root biomass; y-intercept is heterotrophic respiration

Source: Derived from Subke et al. (2006) and Hanson et al. (2000)

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Lumping of soil respiration under heterotrophic and autotrophic respiration also neglects daily and seasonal differences in CO<sub>2</sub> flux as a result of physiological differences among bacteria, fungi and plants. Radiocarbon dating is proving useful (Cisneros-Dozal et al., 2006; Hahn et al., 2006; Schuur and Trumbore, 2006), but there are big differences between results from radiocarbon dating, <sup>13</sup>C labeling and CO<sub>2</sub> efflux studies (Hogberg et al., 2005). New research suggests tight coupling of current photosynthesis with soil respiration, possibly via the supply of labile carbon at the roots (Bond-Lamberty et al., 2004; Sampson et al., 2007; Stoy et al., 2007).

Ectomycorrhizal fungi make up a large proportion of soil biomass and contribute significantly to soil respiration but respond differently to environmental change compared to either roots or bacteria, suggesting a need to separately model bacterial, fungal, and root respiration (Hogberg and Hogberg, 2002; Langley and Hungate, 2003; Fahey et al., 2005; Groenigen et al., 2007; Hogberg et al., 2005; Heinemeyer et al., 2007; Blackwood et al., 2007). Bacterial and fungal, as well as overall faunal community composition affects soil carbon dynamics (Jones and Bradford, 2001; Bradford et al., 2002b; Bradford et al., 2007) but their differing responses may also cancel each other out (Bradford et al., 2002a). Further studies must better clarify understanding of underlying mechanisms and environmental factors that characterize differing microbe responses (e.g. fungi, bacteria) (Chung et al., 2006; Monson et al., 2006; Blackwood et al., 2007; Fierer et al., 2007; Hogberg et al., 2007).

In addition, earthworm effects on carbon and nitrogen cycling are significant (Li et al., 2002; Marhan and Scheu, 2006). Fresh inputs of carbon (i.e. priming) from organisms such as earthworms may allow soil microbes to mine old carbon deeper in the profile (Dijkstra and Cheng, 2007; Fontaine et al., 2007). This suggests that increased input from leaf productivity may boost soil heterotrophic respiration and CO<sub>2</sub> flux from soils. Priming can lead to rapid shifts in community composition (Cleveland et al., 2007; Montano et al., 2007). Low molecular weight compounds, including low molecular weight organic acids, amino acids and sugars, are small products of microbial breakdown, and represent a small fraction of the total mass of carbon cycling through soil. However, breakdown of low molecular weight organic acids may contribute up to 30% of total soil CO<sub>2</sub> efflux because of extremely rapid turnover, with residence times estimated at 1-10 hours (Van Hees et al., 2005).

### **Respiration responses to environmental change**

Under global change scenarios, nitrate deposition from fossil fuel burning, particulate deposition from forest fires, and wind erosion of agricultural soils, are expected to alter forest nutrient cycling. The addition of nitrogen has been shown to affect microbial breakdown of litter and SOM, the results varying with litter type and microbial community composition (Sinsabaugh et al., 2004; Sinsabaugh et al., 2005; Waldrop et al., 2004a). Litter quality is important, at least for temperate forests, where litter in high-lignin systems shows unchanged or decreased rates of decomposition under nitrogen deposition, while low-lignin, low-tannin systems tend to increase decomposition rates (Magill and Aber, 2000; Gallo et al., 2005). In turn, it was shown

in northern temperate forests that the composition of the microbial communities changed in response to nitrogen deposition (Waldrop et al., 2004a). Based on these observations, elucidating carbon dynamics under elevated nitrogen scenarios for other biomes and across canopy tree associations should be a priority.

Fertilization by increased atmospheric CO<sub>2</sub> and the deleterious effects of ozone (O<sub>3</sub>), both resulting from burning of fossil fuels, are also expected to alter forest carbon cycling. In a four-year study in experimental temperate forest stands, Loya et al., (2003) found that a simultaneous 50% increase in CO<sub>2</sub> and O<sub>3</sub> resulted in significantly lower soil carbon formation, possibly due to reduced plant detritus inputs and/or increased consumption of recent carbon by soil microbes. In another study of temperate forest soils, soil faunal communities changed composition under exposure to CO<sub>2</sub> or O<sub>3</sub> singly, but, when combined, there was no main effect (Loranger et al., 2004). Fungal community composition was significantly altered as a response to elevated O<sub>3</sub> in a FACE study in Wisconsin, USA (Chung et al., 2006). The response of fungal respiration to elevated CO<sub>2</sub> is so far equivocal. One study indicated a rise in fungal activity (Phillips et al., 2002) while another recorded a decrease (Groenigen et al., 2007). As in the divergent responses under nitrogen deposition, the result may depend heavily on litter chemical properties. Soil respiration is expected to increase under increased CO<sub>2</sub> and O<sub>3</sub> (Andrews and Schlesinger, 2001; King et al., 2004; Luo et al., 2001b; Pregitzer et al., 2006), but some studies show conflicting results (Suwa et al., 2004; Lichter et al., 2005) or a decrease when combined with fertilization (Butnor et al., 2003). Overall, FACE studies indicate a net increase in carbon storage, mostly in litter and fine root mass, despite soil respiration increases (Allen et al., 2000; Hamilton et al., 2002), although cycling through litter is especially rapid and sequestration in litter is likely limited (Schlesinger and Lichter, 2001).

Respiration response to temperature changes, especially pertaining to a still-hypothetical positive feedback of warming to carbon mineralization, is hotly debated. The early debate centered on whether the soil carbon pool should be lumped or split by rate of turnover (Davidson et al., 2000; Giardina and Ryan, 2000), since (often small) portions of the soil carbon pool cycle very quickly, especially low molecular weight organic acids, and therefore may be more responsive to temperature than larger, older or adsorbed compounds. Later, evidence mounted for an acclimation of soils to heightened temperatures over time, although it now seems clear that depletion of the fast-cycling labile carbon pool under increased initial mineralization rates is partly responsible for the apparent downshifting in respiration over time (Luo et al., 2001a; Melillo et al., 2002; Eliasson et al., 2005). Further experiments are needed to test the acclimation hypothesis. Reworking the data of Giardina and Ryan (2000), others found that the response of the fast pool over experimental scales obscured the slower but ultimately more important response of the large pool of stable carbon (Knorr et al., 2005b). A consensus is still in the making concerning the impact of warming on soil respiration, although it now seems clear that the complex nature of SOC, and confounding factors, including soil water content, complicate a simple determination of the temperature effect (Davidson and Janssens, 2006). Soil respiration is closely coupled to photosynthesis of the canopy, explaining some of the

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apparent causal correlation between temperature and respiration in in-situ studies (Sampson et al., 2007).

The effects of soil freezing, compounded by a decreased or absent snow pack predicted for some temperate and boreal regions, may decrease winter soil respiration (Monson et al., 2006). More attention has been given to drying and wetting cycles recently, which appear to substantially increase annual decomposition (Fierer and Schimel, 2002; Borken et al., 2003; Jarvis et al., 2007).

### STABILIZATION OF CARBON IN FOREST SOILS

Plant-derived organic molecules are stabilized from microbial action by biochemical resistance or by physical protection within soil aggregates or microsites (Table 2 and Figure 6). In order of importance for the stabilization of organic matter in soils across Ohio, USA, drainage class was the only significant determinant of SOM content in the upper 30 cm in forest soils, whereas the significance of individual site variables on SOM content in non-forested soils was firstly soil taxon, then drainage class, and lastly texture. The low significance of these other factors on forest soils suggests different drivers of SOM dynamics in forests (Tan et al., 2004).

The importance of anoxic conditions for preservation of organic matter in boreal peatlands is without doubt (Fierer and Schimel, 2002; Borken et al., 2003; Jarvis et al., 2007). But in aerobic soils, dissolved organic carbon (DOC) that leaches from decomposing material is vulnerable to mineralization and respiration as CO<sub>2</sub> by bacteria or saprophytic fungi. The portion of DOM that escapes mineralization by microbes generally does so by sorption to soil minerals where it is stabilized as SOC. Carbon is also stabilized when fire produces black charcoal from organic matter. Finally, some DOC may be flushed from the soil system during periods of high soil water flow.

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The importance of anoxic conditions for preservation of organic matter in boreal peatlands is without doubt.

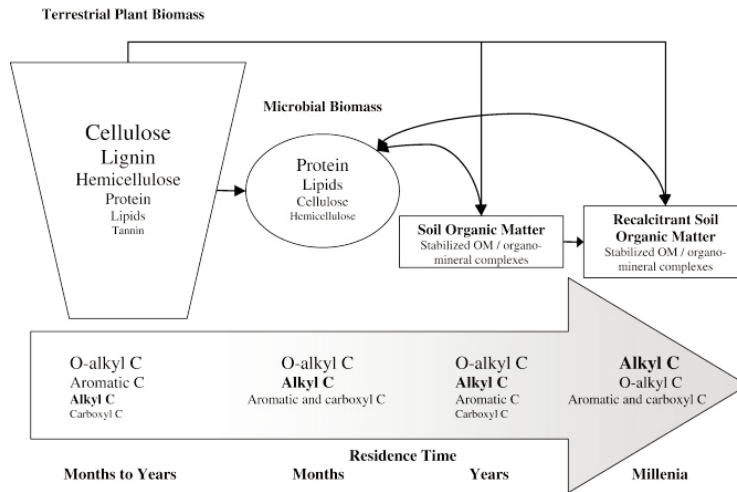
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**Table 2 Mechanisms of carbon immobilization in forest soils.** Some mechanisms are specific to soil order or biome, while others are active in all soils.

<b>Selective preservation</b>	Inherent stability due to e.g. alkyl-C chains in lipids, aromatic structures, phenolics
<b>Spatial segregation</b>	Occlusion inside soil aggregates
	Sequestration within soil micropores
	Coating with hydrophobic aliphatic compounds
	Intercalation within phyllosilicates (clay)
<b>Molecular interaction</b>	Complexation with metal ions
	Interaction with other organic molecules through ligand exchange, polyvalent cation bridges or weak interactions
<b>Inundation</b>	Anoxic conditions prevent abiotic oxidation and aerobic microbial respiration
<b>Freezing</b>	Sub-freezing temperature stifles microbial respiration
<b>Carbonization</b>	Relatively inert carbon is broken down only at millennial timescales

Source: After Lorenz et al., (2007) and Lutzow et al., (2006)

**Figure 6 Carbon flux through terrestrial organic matter pools (top) and relative enrichment of recalcitrant alkyl carbon during breakdown of more-labile compounds (bottom).** The relative composition of organic matter by carbon structure is represented by size of type.



Source: Reprinted from Lorenz, K., Lal, R., Preston, C.M., Nierop, K.G.J., 2007. Strengthening the soil organic carbon pool by increasing contributions from recalcitrant aliphatic bio(macro)molecules. *Geoderma* 142, 1-10, with permission from Elsevier.

### Sorption and complexation of dissolved organic matter

In aerobic soils, texture is considered to be the most important driver of DOC stabilization in soil, with mineralogy an important factor that is dependent on texture (Bird et al., 2002). Across a 1,000 km latitudinal transect in Siberia, SOC stocks on fine-textured soils were approximately double the stocks on coarse-textured soils (Bird et al., 2002). The layered structure of clay results in an extremely high surface area to volume ratio, and clay interlayers host a multitude of cations which provide binding sites for DOC. Clay content exerts a powerful control on the size of the older soil carbon pool in Amazonian soils (Telles et al., 2003). However, Giardina et al., (2001) found no relation between carbon mineralization rates and clay content in laboratory-incubated upland forest soils. Clay content of soil is well-correlated with SOC generally, although other factors dominate DOC stabilization in cold or wet climates. Sorption to the mineral matrix has been shown to strongly preserve DOM (Kaiser and Guggenberger, 2000). Aluminum (Al) and iron (Fe) cations are the most important interlayer mineral binders for DOM (Zinn et al., 2007). Besides binding to clay particles, colloidal and soluble organic matter can form insoluble complexes with Al and Fe cations, which precipitate (Schwesig et al., 2003; Rasmussen et al., 2006; Scheel et al., 2007). These results suggest that whole-ecosystem carbon cycle models should account for both soil texture and soil mineralogy when modeling carbon fluxes (Table 3). Labile DOM high in carbohydrate has a large increase in stability due to sorption, but for DOM with a greater proportion in complex aromatic organic compounds stability due to sorption is relatively small because such compounds are

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In aerobic soils, texture is considered to be the most important driver of DOC stabilization in soil, with mineralogy an important factor that is dependent on texture.

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already relatively stable. However, irrespective of proportional increases, gross sorption of recalcitrant compounds was much larger than sorption of labile compounds, in fact as much as four times larger (Kalbitz et al., 2005). Stabilization of OM by sorption therefore depends on particulars of the organic compounds sorbed, strong chemical bonds to the mineral soil, and/or a physical inaccessibility of OM to microorganisms (Kalbitz et al., 2005).

**Table 3 Characteristics of six process-based forest soil carbon models**

	<b>Yasso</b>	<b>ROMUL</b>	<b>SOILN</b>	<b>RothC</b>	<b>Forest-DNDC</b>	<b>CENTURY</b>
<b>Time-step</b>	Year	Month	Day	Month	Day	Month
<b>Simulation depth</b>	Organic layer +1 m mineral soil	Organic layer +1 m mineral soil	Any depth	Adjustable from 0 to 1 m	Adjustable from 1 to 1.5 m	20 cm
<b>Organic matter pools</b>						
<b>Stand</b>	-	-	roots, stems, leaves, grains	-	canopy, understory, groundstory	leaves, fine roots, fine branches, coarse wood, coarse roots
<b>Litter</b>	fine and coarse woody litter	aboveground and belowground pools divided by N and ash contents	1-2 per soil layer, 10-15 soil layers	resistant and decomposable pools	very labile, labile, and resistant pools for each soil layer	aboveground and belowground pools divided into metabolic and structural, senescent litter pooled based on lignin:N ratio
<b>Soil</b>	extractives, celluloses, lignin-like compounds, 2 humus pools	six or more	1 humus, 1 microbe pool per soil layer, 10-15 soil layers	living, humic and insoluble OM pools	2 humads and humus per layer	active, slow and passive SOM pools
<b>Different pools for organic and mineral soil?</b>	No	Yes	Yes	No	Yes	No
<b>Nutrient input</b>	-	N deposition	N deposition, fertilization, N content of plant parts	-	N deposition and fertilization	N deposition and fertilization, organic N inputs, P, S
<b>Soil texture input</b>	-	Clay content	Hydraulic properties	Clay content	Clay content	Sand, silt and clay content
<b>Limitations</b>	Upland forest soils only	Well or excessively drained mineral soils only	Substantial input information required, not for peatlands	Upland forest soils only	Substantial input information required	Very shallow, not for peatlands, does not separate humified litter from mineral soil
<b>Measurability of pools</b>	Only extractives, celluloses and the sum of the other pools are measurable	Yes, all pools measurable	No, pools are conceptual and cannot be directly measured	Yes, all pools measurable	No, pools are conceptual and cannot be directly measured	SOC, litter pools measurable, sub-pools conceptual only

Source: Adapted from Peltoniemi et al. (2007)

### Fire as sequestration mechanism

The many effects of fire on forest soils have been reviewed by Certini (2005). In areas with frequent fires, 35-40% of SOC was fire-derived black carbon. Fire can sterilize soils to depths of 10 cm or more, and effects of sterilization may last a decade, resulting in decreased microbial respiration. When fire does not remove carbon from the soil system through combustion, it tends to increase the stability of the carbon remaining through carbonization, reduction in water solubility, and relative enrichment in aromatic groups (Certini, 2005). Workers in a boreal Siberian Scots pine forest found that black carbon contributed a small percentage of the SOC pool while the fire reduced the mass of the forest floor by 60% (Czimczik et al., 2003). A wildfire in boreal Alaska burned polysaccharide-derived compounds preferentially, resulting in a relative enrichment of lipid- and lignin-derived compounds (Neff et al., 2005). There appears to be an inverse relationship between fire frequency and

complete combustion: infrequent fire return intervals and high intensity may result in less carbonization and more complete combustion than in regions with shorter fire return intervals that experience lower-intensity fires, increased carbonization, and therefore increased storage (Czimczik et al., 2005).

## QUANTIFYING THE CARBON UNDER FORESTS

In the northern hemisphere, the carbon in soils remains the highest uncertainty in global budgeting (Goodale et al., 2002) and partitioning soil respiration among sources to identify carbon leakage/loss has proved to be one of the most difficult tasks (Ryan and Law, 2005). Failure to close the soil carbon budget stems from discrepancies between measured bulk CO<sub>2</sub> fluxes and the predictions of process models of autotrophic and heterotrophic respiration (Trumbore, 2006). Different methods used to accommodate study objectives and resources make comparison difficult (Wayson et al., 2006) and as a result, many budgets leave out soil carbon and litter carbon accumulations completely (Liski et al., 2003)

### Quantifying carbon additions

The turnover of organic matter in surface soil layers can be quantified by direct measurement of mass loss through litterbag studies or by <sup>14</sup>C enrichment, litter sampling and mass spectrometer analysis. However, there are known problems with both litterbag studies and <sup>14</sup>C enrichment as methodologies for measuring carbon addition to soils. Litterbags limit breakdown of litter by soil macrofauna. The <sup>14</sup>C signature measures the mean residence time of carbon in the surface layer, but not the lifetime of various recognizable litter components (e.g. from fine roots, leaves, bark) (Hanson et al., 2005). And it has been found that <sup>14</sup>C-labeled carbon residence time in fine roots, estimated at >4 years, is much longer than the <1 year root longevity estimated by using the minirhizotron, a small camera lowered through a clear plastic tube to monitor the growth of roots over time (Strand et al., 2008).

It was previously thought that radiocarbon dating and the turnover time for roots – estimated by laborious sorting and weighing of root production year-to-year and then dividing total root biomass by annual production – could be reconciled if the age distribution of roots were positively skewed (Tierney and Fahey, 2002; Trumbore and Gaudinski, 2003). The most current thinking on important sources of discrepancy between direct root weighing in measuring fine root turnover are outlined by Strand et al., (2008) and include: 1) the presence of different root pools cycling at different rates; 2) the confounding effect of stored carbohydrates, which would throw off radiocarbon estimates of age; 3) the skewed nature of root age distribution as pointed out in Trumbore and Gaudinski (2003); 4) lingering effects of minirhizotron installation on root growth; and 5) the use of median root longevity as an inaccurate substitute for mean longevity in minirhizotron studies. These sources of error cause radiocarbon methods to underestimate the importance of fine root turnover to soil carbon cycling and the minirhizotron method to overestimate this

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In the northern hemisphere, the carbon in soils remains the highest uncertainty in global budgeting

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importance (Strand et al., 2008). Work is underway to address these shortcomings and to deepen understanding of root turnover, e.g. by partitioning root pools by branching order (Guo et al., 2008).

### **Partitioning soil respiration**

The major classes of soil partitioning techniques and a detailed discussion of each are outlined by Hanson et al., (2000) (Table 1). The isotope dating approach and conventional carbon sorting for estimates of root turnover currently cannot be reconciled (Hogberg et al., 2005). Radiocarbon measurements of fine root turnover are biased by the presence of non-structural carbohydrate reserves (Luo, 2003). Minirhizotron measurement of fine root turnover cannot be compared to other methods until an objective determination of fine root biomass can be made (Vogt et al., 1998). As mentioned previously, root age is positively skewed, causing the minirhizotron method to systematically overestimate root turnover, and the radiocarbon tracer method to systematically underestimate turnover (Tierney and Fahey, 2002). A comprehensive review of research needs in measuring and modeling soil respiration has been done by Ryan and Law (2005), while Subke et al., (2006) provide an exhaustive list of soil CO<sub>2</sub> efflux partitioning studies through 2006 across all terrestrial biomes. They show that many of the techniques for partitioning have inherent methodological biases (Subke et al., 2006). For example, detection of changes, especially depletion, of the large, slow-cycling pool of recalcitrant soil carbon represents a significant challenge and is almost always underrepresented, but it is essential to quantifying the carbon exchange between soil and the atmosphere. Additionally, multi-factor experiments must be of sufficient length to allow adjustment to treatment conditions (Ryan and Law, 2005). It is therefore better to resample the same points than to randomly select new ones in long-term sampling studies and inventories. Due to the spatial variability of SOC processes, 15%-20% changes in soil carbon stocks may be overlooked (Yanai et al., 2003). Site variability therefore confounds broad-scale application of flux data (Hibbard et al., 2005). There is also a trade-off between spatial and temporal resolution when using manual vs. automated CO<sub>2</sub> flux measurements. Used in combination, the two systems provide combined resolution in both dimensions, but manual measurements are sufficient for measuring integrated seasonal fluxes (Savage and Davidson, 2003).

### **Modeling soil carbon dynamics**

Many SOC models have been created for agricultural systems, and may be modified to simulate forested systems, in order to accommodate important differences in management, disturbance regime, vegetation, and biota. To date there has been only one published model comparison for forest SOC dynamics. Peltoniemi et al., (2007) review and compare six process-based, multiple-SOC-pool models of forest SOC dynamics. The review includes an extensive comparison of model inputs and modeled processes. More work is needed to assess the accuracy of forest soil carbon models, and to adapt them or develop new ones for diverse biomes. Critically, very few countries and regions have published long-term soil carbon datasets with the

ancillary data needed to verify model accuracy, and there is a total lack of such inventories for tropical regions. Most soil carbon process models do not deal explicitly with peatlands, severely limiting their applicability in some boreal regions. Wetlands versions of the *RothC* and *ROMUL* models are expected in the near future, and *Forest-DNDC* includes a wetland component (Peltoniemi et al., 2007). The move toward process-based models of SOC dynamics is hindered by poor understanding of the different mechanisms of sequestration of diverse classes of organic biomolecules in soils. Model soil carbon pools must be derived from functional classes of compounds (which must first be characterized) with similar sequestration mechanisms, rather than from the most easily differentiated classes of SOC based on *in situ* measurement techniques or fractionation (Lutzow et al., 2006) (Table 3).

Empirical models must be careful to parameterize at the same time step as the output (Janssens and Pilegaard, 2003). The concentration of labile carbon, its rapid turnover, and the resultant large CO<sub>2</sub> efflux can obscure the sensitivity of heterotrophic respiration to soil temperature change. Care should be taken to control for labile carbon concentrations when extrapolating field measurements of bulk soil respiration to global change scenarios (Gu et al., 2004).

The temperature dependence of soil is often described by the Q<sub>10</sub> value, which is defined by the difference in respiration rates over a 10° C interval. Q<sub>10</sub> has been found to be extremely variable, with a range from 1 (no effect of temperature on respiration) to 5 (five times higher respiration rate with a 10 degree rise in temperature) under different combinations of soil moisture and soil temperature (Reichstein et al., 2003). Kinetic properties of the many organic compounds in soils, plus environmental constraints such as limiting soil moisture or nutrients, complicate efforts to fully explain the temperature sensitivity of microbial respiration (Davidson and Janssens, 2006). In an analysis of sources of uncertainty in the soil carbon model *SWIM*, Post et al., (2008) identified the carbon mineralization rate, carbon use efficiency, Q<sub>10</sub>, soil bulk density, and initial carbon content as the most critically sensitive parameters. Better models will have to differentiate the direct effects of drying, wetting, and carbon substrate supply to soil microbes from the indirect effects of soil water content and temperature on diffusion of carbon substrates to the microbial population (Davidson et al., 2006). Work in this area indicates that models incorporating realistic spatial relationships, hourly time steps, and mechanistic workings give the most accurate results (Hanson et al., 2004). Not all applications will be suited to process models, however, due to the extensive inputs required (Liski et al., 2005).

### The superficial nature of soil carbon research

Studies of soil organic matter under conventional and no-till soil management in agriculture have been largely limited to the top 30 cm of soil. Now, some are suggesting the need to consider SOM deeper in the profile (Baker et al., 2007). The same argument ought to be made for forest soil research: rooting depths are far greater for many tree species than field crops. Soil depth confounds warming studies by insulating deeper soil layers (Pavelka et al., 2007) and delaying CO<sub>2</sub> efflux (Jassal

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Studies of soil organic matter under conventional and no-till soil management in agriculture have been largely limited to the top 30 cm of soil.

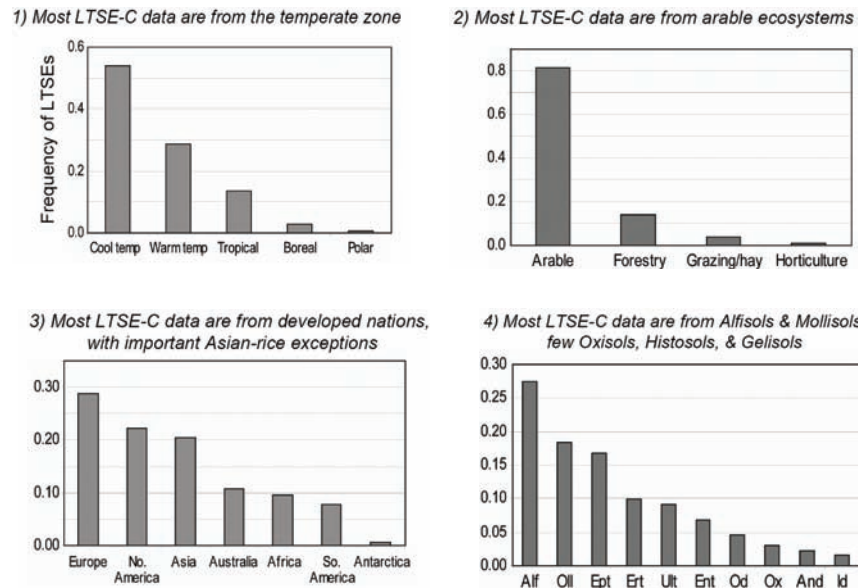
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et al., 2004; Drewitt et al., 2005). Also, a significant portion of below-ground carbon is deeper than 1 m (Jobbagy and Jackson, 2000) and recent research indicates that roots exert powerful influences on redox activity in their vicinity, with important implications for carbon cycling deep in the soil profile (Fimmen et al., 2008).

**Quantifying carbon stocks after land use change**

Long-term soil experiments and inventories can elucidate SOC dynamics in ways that shorter ones cannot. Peltoniemi et al. (2007) point out the importance of repeated soil surveys for SOC model verification and validation. There is currently no unified global network of long-term soil experiments (LTSEs), despite the importance of chronosequence studies for area-based carbon budgeting under land use change (Woodbury et al., 2007). A lack of uniformity of measurements complicates comparison and synthesis. There are other problems: only 20% of soil studies measuring SOC are in forested biomes; therefore, boreal, tropical and warm-temperate forests are underrepresented; soil studies measuring SOC are heavily concentrated in developed countries; and long-term SOC studies on alfisols and mollisols dominate, while long-term changes on oxisols, histosols and gelisols are still poorly understood (Figure 7). Chronosequences, or space-for-time substitutions, though useful for characterizing soil change over centuries or millennia, may confuse the effects of land use with weathering. Land use history can be difficult to properly control for (Richter et al., 2007) (Table 4).

**Figure 7** Distribution of long term soil experiments (LTSE) measuring SOC across climate zones, land uses, continents and soil orders. Note the lack of forest LTSEs despite the importance of land use change, specifically deforestation, to national and global carbon budgets.



Source: Graphs produced by D.D. Richter from data at <http://ltse.env.duke.edu>. Used with permission.

**Table 4** Types of soil experiments that may be used to elucidate carbon dynamics and changes in carbon stocks under land-use change

Approach	Time scale	Uses and strengths	Challenges and limitations
	yr		
Short-term soil experiments	<1–10	field or lab based, experimental control, versatile, short-term processes	extrapolation to larger scales of space and time, reductionist
Long-term soil experiments	>10	field based, direct soil observation, experimental control, sample archive	duration before useful data, vulnerable to loss or neglect, extrapolation to larger scales
Repeated soil surveys	>10	field based, direct soil observation, regional perspective, sample archive	planning and operational details, very few yet conducted, monitoring
Space-for-time- substitution	>10 to >>1000	field based, highly time efficient	space and time confounded
Computer models	<1 to >>1000	versatile, heuristic and predictive, positively interact with all approaches	dependent on observational data

Source: Richter, D.D., Hofmockel, M., Callahan Jr, M.A., Powlson, D.S., Smith, P., 2007. Long-term soil experiments: Keys to managing Earth's rapidly changing ecosystems. *Soil Science Society of America Journal* 71, 266-279. Reprinted with permission.

### EFFECTS OF MANAGEMENT REGIME ON SOIL CARBON CYCLING

The Fourth Assessment Report by the IPCC Working Group III projects that, initially, reduction in deforestation will lead to the greatest positive increase in global carbon sequestration, due to the current rapid rate of deforestation and the large associated CO<sub>2</sub> loss to the atmosphere. Over the long term, sustainable forest management that increases forest growing stock while also providing timber, fiber and energy will provide the greatest mitigation benefit at the lowest cost to society (IPCC, 2007). But the link between different forest management activities, deforestation, reforestation and afforestation and the net carbon flux between soils and the atmosphere is not well characterized (Table 5).

Productivity of the forest increases litter fall and sequestration; less disturbance of soil tends to preserve soil carbon stocks; and mixed species forests are more resilient and therefore better systems for securing carbon in forest soils. On the other hand, planting on agricultural soils increases carbon uptake by soils for both conifers and broadleaf trees (Morris et al., 2007). Although the rate of carbon accumulation and sequestration within the soil profile differs by tree species, no species effect on SOM stability has yet been reported (Jandl et al., 2007). Differences in plant anatomy lead to changes in the vertical distribution of minerals and soil carbon when there is land use or land cover change (Jackson et al., 2000). For example, in Fujian, China, conversion of natural forests to plantations has been linked to carbon loss (Yang et al., 2007). However, combined CO<sub>2</sub> sequestration and timber production can be economically maximized (Thornley and Cannell, 2000). In addition, during reforestation, soils are a slower but more persistent sink than aboveground carbon, and are more stable pools than aboveground pools for actively harvested forests (Thuille et al., 2000).

Studies in boreal forests have demonstrated that tree harvesting generally has little long-term effect on stable soil carbon stocks (Martin et al., 2005), although evidence from temperate forests of the southeastern USA indicates that whole tree harvesting can be detrimental by removing nitrogen (Johnson et al., 2002). There are no obvious long-term effects from clearcutting that can be detected on in-stream DOC chemistry (Johnson et al., 2002), although clearcutting released a pulse of labile DOC at Hubbard

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Studies in boreal forests have demonstrated that tree harvesting generally has little long-term effect on stable soil carbon stocks.

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Brook in New Hampshire, USA (Dai et al., 2001), probably from humic substances at the forest floor (Ussiri and Johnson, 2007). Shortened rotations from 90 to 60 years in Finland increased soil carbon by increasing input of litter but did not maximize system-wide carbon sequestration because of increased frequency of harvest operations (Liski et al., 2001); although others have found that fresh carbon additions due to harvesting operations can stimulate microbial populations to mineralize ancient deep soil carbon (Fontaine et al., 2007; Jandl et al., 2007) (Figure 8).

**Table 5** The generalized impact of forest management actions on carbon stocks

Summary of the effects of specific forest management actions on ecosystem C stocks ('+'...increases C stock, '-'...decreases C stock; '±' neutral with respect to C stock)

*Afforestation*

- + Accumulation of aboveground biomass formation of a C-rich litter layer and slow build-up of the C pool in the mineral soil
- ± Stand stability depends on the mixture of tree species
- Monotone landscape, in the case of even-aged mono-species plantations

*Tree species*

- + Affects stand stability and resilience against disturbances; effect applies for entire rotation period; positive side-effect on landscape diversity, when mixed species stands are established
- Effect on C storage in stable soil pools controversial and so far insufficiently proven

*Stand management*

- + Long rotation period ensures less disturbance due to harvesting, many forest operations aim at increased stand stability, every measure that increases ecosystem stability against disturbance
- ± Different conclusions on the effect of harvesting, depending if harvest residues are counted as a C loss or a C input to the soil
- Forests are already C-rich ecosystems — small increase in C possible; thinning increases stand stability at the expense of the C pool size; harvesting invariably exports C

*Disturbance*

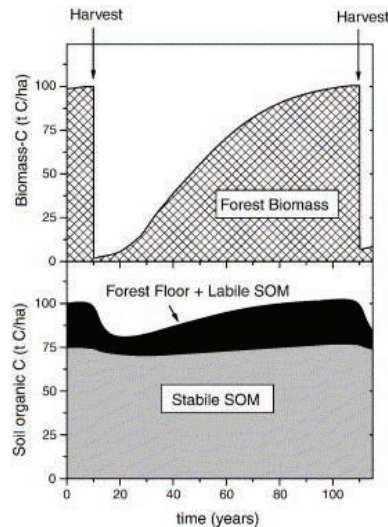
- + Effects such as pest infestation and fire can be controlled to a certain extent
- ± Low intensity fires limit the risk of catastrophic events
- Catastrophic (singular) events cannot be controlled; probability of disturbance can rise under changed climatic conditions, when stands are poorly adapted

*Site improvement*

- + N fertilization affects aboveground biomass; effect on soil C depends on interaction of litter production by trees and carbon use efficiency of soil microbes
- ± Drainage of peatland enables the establishment of forests (increased C storage in the biomass) and decreases CH<sub>4</sub> emissions from soil, but is linked to the increased release of CO<sub>2</sub> and N<sub>2</sub>O from the soil
- Liming and site preparation always stimulate soil microbial activity. The intended effect of activating the nutrient cycle is adverse to C sequestration; N fertilization leads to emission of potent greenhouse gases from soils

Source: From Jandl, R., et al. 2007. How strongly can forest management influence soil carbon sequestration? *Geoderma*, 137: 253–268. Reprinted with permission.

**Figure 8** A simulation of carbon stocks above- and belowground before and after forest harvesting, for a typical Central European Norway spruce forest. Assumptions include a 100 year rotation for a typical Norway spruce stand with 25% labile SOM.



Source: Reprinted from Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D.W., Minkinen, K., and Byrne, K.A. 2007. How strongly can forest management influence soil carbon sequestration? *Geoderma*, 137: 253–268. Reprinted with permission.

Plant diversity and composition effects on net primary productivity (NPP) are becoming apparent and must be accounted for (Catovsky et al., 2002). Oak forests turn SOM over faster compared to pine, which locks up more litter for longer in the surface layers (Quideau et al., 2001). Broadleaf tree plantations replacing natural forest or pasture tend not to change soil carbon stocks, while pine plantations reduce soil carbon stocks 12-15% (Guo and Gifford, 2002). Conversion of forest to pasture results in a slow but marked increase in soil carbon stocks, but this is the reverse for tilled agriculture (Cerri et al., 2003; Cerri et al., 2004). Pasture systems are very productive and thus larger carbon fluxes from them indicate greater allocation of carbon belowground (Johnson and Curtis, 2001; Johnson et al., 2002; Paul et al., 2002; Salimon et al., 2004; Thuille and Schulze, 2006).

## CONCLUSION AND SUMMARY RECOMMENDATIONS

This review outlined the most critical issues and impediments to characterizing belowground carbon cycling in forested biomes. To further our understanding of belowground carbon dynamics in forests, more work is needed to characterize the following:

- controls on the depth of the forest floor organic layer by leaching of dissolved organic carbon (DOC) to the mineral soil.

- rates of fine root turnover across species and biomes.
- patterns of bacterial, fungal and plant respiration and responses to physical and biotic forcing.
- dynamics of functionally-distinct soil carbon pools, rather than the most easily measured and fractionated pools.
- the most accurate methods for quantifying forest soil carbon stocks and fluxes.

The global nature of the carbon cycle requires a globally-distributed and coordinated research program, but has thus far been largely limited to:

- the developed world
- the top 30 cm of the soil profile
- temperate biomes
- agricultural soils

Political and financial resources are being mobilized to increase the stock of carbon in forest soils despite minimal research to date about the long-term effects of land use on SOC stocks.

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## Chapter 3

# The Physiological Ecology of Carbon Science in Forest Stands

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### EXECUTIVE SUMMARY

In order to better understand the ways in which future forests will change and be changed by shifting climates, it is necessary to understand the underlying drivers of forest development and the ways these drivers are affected by changes in atmospheric carbon dioxide concentrations, temperature, precipitation, and nutrient levels. Successional forces lead to somewhat predictable changes in forest stands throughout the world. These changes can lead to corresponding shifts in the dynamics of carbon uptake, storage, and release.

Many studies have attempted to elucidate the effects of changing climate conditions on forest ecosystem dynamics; however, the complexity of forest systems, long time horizons, and high costs associated with large-scale research have limited the ability of scientists to make reliable predictions about future changes in forest carbon flux at the global scale. Free Air Carbon Dioxide Enrichment (FACE) experiments are suggesting that forest net primary productivity, and thus carbon uptake, usually increases when atmospheric carbon dioxide levels increase, likely due to factors such as increased nitrogen use efficiency and competitive advantages of shade tolerant species. Experiments dealing with drought and temperature change are providing evidence that water availability, especially soil moisture, may be the most important factor driving forest carbon dynamics. Forest ecosystem experiments, such as FACE programs, have not been operating long enough to predict long term responses of forest ecosystems to increases in carbon dioxide. The expense and time constraints of field experiments force scientists to rely on multifactor models (the majority of which account for five or fewer variables) leading to results based on large assumptions.

If predictions are made regarding stand level carbon within forest ecosystems, it is important to have an understanding of what scientific research has or has not

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established. Key findings of this review summarize what we do and do not know about stand dynamics and carbon.

### **What we do know about stand dynamics and carbon assimilation**

- Forests have relatively predictable stages of development that have been termed initiation, stem exclusion, understory initiation, and old growth.
- The nature of type, scale, and frequency of disturbances and their effects on forests are well documented and their effects on the nature of the origin of new or released regeneration well understood.
- Most studies support the notion that the stem exclusion stage is a period of high carbon assimilation, water uptake, and nutrient acquisition.
- Recent studies are showing that old growth forests are not just storing carbon, but are also sequestering significant amounts – particularly in large tropical basins such as the Congo and the Amazon.
- Free Air Carbon Dioxide Enrichment experiments (FACE) have provided insights into our understanding of the physiological and stand level responses (+/- feedbacks) to elevated carbon dioxide over short periods of time (15 years). Stands in the stem exclusion stage are expected to increase sequestration, with increase in water use and nutrient use efficiencies, and a potential to favor shade tolerant species.
- Stand level rainfall exclusion and addition experiments have provided insight into carbon reallocation, carbon respiration and storage processes, drought aversion and avoidance adaptations, and +/- feedbacks with other soil resources (e.g. soil fertility). Results suggest that timing of drought (growing versus non-growing season) and species composition change are two factors to consider.

### **What we do not know about stand dynamics and carbon assimilation**

- Although we understand the stages of stand development, there is considerable unpredictability in the actual nature of species composition, stocking, and rates of development because of numerous positive and negative feedbacks that make precise understanding of future stand development difficult.
- Carbon stocks and fluxes across and within different forest biomes – particularly in the tropics – have not been well documented.
- While informative, FACE experiments are limited to temperate stands that are mostly in the stem exclusion stage – only some of this information can be applied to tropical regions and other developmental stages.
- More studies are needed that investigate the multiple interactions of limiting and non-limiting resources of soil nutrients, soil water availability, and temperature fluctuations in elevated carbon dioxide environments.

**Keywords:** carbon flux, forest dynamics, FACE, sequestration, disturbance, climate change, carbon balance, warming, precipitation, NPP, succession

## INTRODUCTION

Understanding how future forests will affect and be affected by changing climates requires an understanding of the principles governing the development of forests over time. In an effort to provide a comprehensive understanding of stand level changes in forest carbon with relation to climatic conditions, we present a synthesis of the literature.

Although there are many forest types composed of seemingly infinite combinations of species, similarities in stand development produce analogous stand structures in most of the world's forests (Oliver, 1992). Successional processes alter both forest structure and accompanying ecological processes in predictable ways (Cowles, 1911; Odum, 1969, 1971; Shugart and West, 1980; Bormann and Likens, 1979; Oliver, 1981; Hibbs, 1983; Glenn-Lewin et al., 1992; Oliver and Larson, 1996; Barnes et al., 1998) and regulate changes in forest biomass (Odum, 1969) in systems as seemingly disparate as the tropical rainforests of the Amazon and the boreal forests of the Canadian Shield (Oliver, 1992). The amount of carbon within a forest stand is a factor of both forest structure and competition between individuals.

In this chapter we first describe the concept of stand dynamics, the stages of stand development, and their relevance to our understanding of carbon assimilation and storage in forests. We then describe the physiological processes of photosynthesis and carbon dioxide assimilation, and the effects of other limiting resources on this process (soil water availability, soil nutrients). We then describe the experimental approaches used to manipulate resources (soil, water, air) and monitor such effects on stand developmental and physiological processes – especially carbon assimilation and storage. In this section we describe the Free Air Carbon Dioxide Enrichment (FACE) experiments with a review of the results so far and their limitations. We also describe several other stand-scale experiments that have manipulated precipitation – another key climate effect on forests. We then conclude with summary recommendations on further work that is needed.

## THE CONCEPT OF STAND DYNAMICS

Relatively predictable changes in forest stand structures over time occur in continuous sequential stages (Bormann and Likens, 1979; Oliver, 1981; Peet and Christensen, 1987; Oliver and Larson, 1996; Franklin et al., 2002) which various authors have described using differing terminology. However, they all outline a progressive shift in community dynamics from colonization to competition to peak growth and then slow decay (Figure 1).

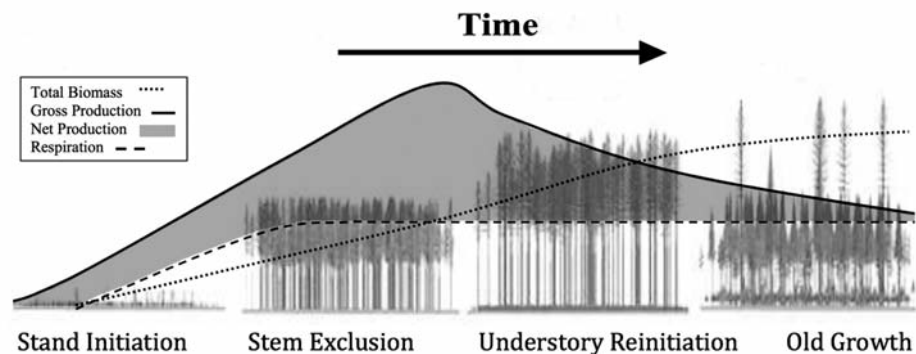
The four stages of stand dynamics as described by Oliver and Larson (1996) are:

1. **Stand initiation** takes place following disturbance and is usually characterized by large numbers of young trees growing from seed or sprouts to rapidly occupy newly available growing space. This period of invasion is critical in determining the trajectory of a developing stand. During this stage the environment in the stand transforms relatively quickly as the

influence of re-vegetation, site parameters, disturbance type, and the return to biogeochemical balance all shape the rapidly developing stand (Bormann and Likens, 1979; Canham and Marks, 1985).

2. **Stem exclusion** is the period of intense competition for resources (e.g. light, soil water, nutrients) and for physical space, characterized by high rates of mortality and rapid assimilation of nutrients and carbon. Maximum assimilation rates of carbon and biomass occur during this stage.
3. **Understory initiation** begins as the survivors of stem exclusion grow older and weaken in resource acquisition. The remaining trees are not able to fully utilize the released growing space and new cohorts establish in the understory.
4. **Old growth** follows as the overstory trees of the initiating cohort die, breaking the uniformity of the canopy, allowing for their slow replacement by the new cohorts established during and after understory initiation. This process leads to the characteristic presence of multiple cohorts. This stage has foliage distributed throughout the vertical layers of the canopy with “horizontal heterogeneity, often evident as canopy gaps and dense reproduction patches” (Franklin and Van Pelt, 2004).

**Figure 1 Stand dynamics, respiration, production, and total biomass.** As stands age they move through predictable stages of development, with predictable consequences for production. Stand level carbon stocks in the form of biomass and coarse woody material increase as a stand progresses through successional stages. The rate of increase in biomass is also not constant over the life of a stand; early stages of stand development have low rates of biomass accumulation due to trees re-establishing themselves on the site. During stand initiation, net production steadily increases and peaks during the stem exclusion stage. Different stands will move through these stages at different rates influenced by species composition, climate, disturbance and other site factors. Stands with the same species composition growing on favorable sites will not only accumulate carbon at a higher maximum rate but they will also reach this maximum rate sooner than stands on poor sites.



Source: Adapted from Oliver and Larson (1996)

Forests move through these successional stages at varying rates and along a multitude of possible trajectories depending on stem density (competition), species composition and available resources (site factors), climate, disturbance patterns and human activity.

Growth and uptake of carbon are quantified in different ways. When comparing growth activity in different ecosystems and stand structures it is important to use the same measurement and methodological approach. Measures of net primary production (NPP) and net ecosystem production (NEP) are used to quantify ecosystem uptake of carbon. NPP is the overall net uptake of carbon by primary producers (organisms that photosynthesize) in an ecosystem per unit of time. NEP is the overall net uptake or release of carbon by an ecosystem per unit of time. Ecosystems are often stratified and NPP and NEP can refer to all or just part of an ecosystem. Biomass is another way of monitoring change in forest stands; it is the mass of organic matter in an ecosystem. Biomass can be stratified into many groups including, but not limited to: living biomass, woody biomass, and above and below ground biomass. The importance of understanding what measure is being used to quantify carbon, and for what part of an ecosystem, cannot be stressed enough, because confusing these will lead to false conclusions.

### THE PHYSIOLOGY OF TREES AND FOREST STANDS

Trees and other vegetation can uptake and sequester atmospheric carbon and draw up moisture from the soil, transpiring it to the atmosphere, profoundly influencing climate (Chapin III et al., 2002) (Figure 2).

Understanding the basic physiological processes of photosynthesis and transpiration is essential if reliable assumptions are to be made about the effects of elevated CO<sub>2</sub> on future forests (Long, 1998). Individual trees share similarities at both the micro and macro level in physiology, morphology, requirements for survival and patterns of photosynthate allocation. Although they are more complex in both structure and function than other plants, they are physiologically similar (Oliver, 1992; Oliver and Larson, 1996).

All carbon allocation in plants – and subsequently trees – can be divided into three categories:

- Respiration, both for the ongoing maintenance of tissues and for the synthesis of compounds used in the growth of new tissues.
- Vegetative growth of roots, stems, and leaves.
- Reproductive growth used to produce flowers, cones, fruit, and seeds.

The relative carbon allocation priorities vary from species to species, with age, by stage of stand development, and with biotic, edaphic, climatic, and physiographic site factors (Grime, 1977; Keyes and Grier, 1981; Tritton and Hornbeck, 1982; Dickson, 1986; Ericsson et al., 1996; Lacoïnte, 2000; Gower et al., 2001; Larcher, 2003; Lockhart et al., 2008). The complexities inherent in this shifting priority have been known to plant physiologists for some time and have been well demonstrated in trees. Factors such as the availability of light, water and nutrients, atmospheric CO<sub>2</sub> concentrations, or variations in temperature can lead to significant changes in both the proportional allocation and the total rate of fixation of carbon in trees (Aber et al., 1985; Chapin

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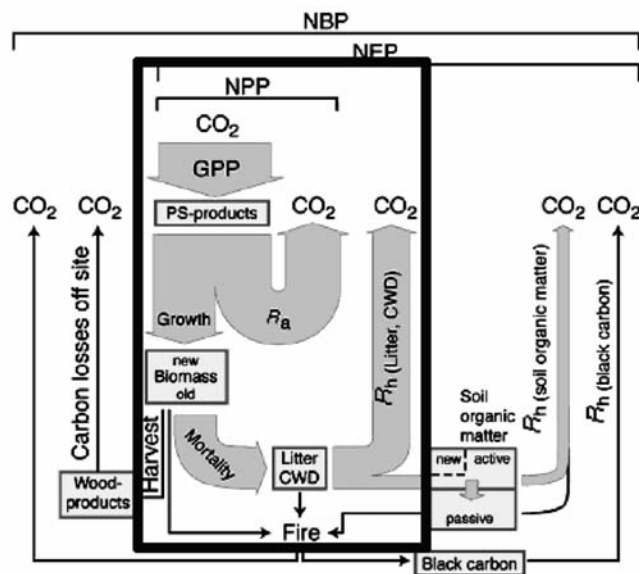
The importance of understanding what measure is being used to quantify carbon, and for what part of an ecosystem, cannot be stressed enough, because confusing these will lead to false conclusions.

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III et al., 1987; Steeves and Sussex, 1989). The effect of any one or any combination of these factors on carbon uptake is predictable (Farrar, 1999), but the magnitude of the effect varies greatly both between and within species (Raghavendra, 1991). For example, in a temperate forest in North Carolina, winged elm (*Ulmus alata*) regeneration had a 21% relative increase in growth under elevated carbon dioxide while black locust (*Robinia pseudoacacia*) had a 230% relative increase under the same amount of carbon dioxide elevation (Mohan et al., 2007). While predicting how environmental changes may affect a single tree is challenging on its own, estimating the effects of similar changes at the stand or landscape scale is extremely difficult (Lavigne, 1992; Schulze, 2000).

**Figure 2 Forest carbon flux.** Half of all carbon dioxide absorbed by forests is used for respiration maintenance; the remainder is stored as biomass. Branch, leaf and root turnover, eventual tree death, and inevitable decomposition transfer carbon back to the atmosphere or into the soil carbon pool.



Source: Schulze, E. D., Wirth, C., Heimann, M., 2000. Managing forests after Kyoto. *Science* 289, 2058-2059. Reprinted with permission from AAAS.

Plants take in the CO<sub>2</sub> necessary for photosynthesis by opening leaf stomata, allowing access to the gas exchange sites located in the mesophyll (Larcher, 2003). In doing so, they also transpire moisture into the atmosphere. Plants, therefore, face a delicate balance between the loss of water – forcing the expenditure of energy to replace it – and the need for the CO<sub>2</sub> necessary to fuel photosynthesis. The demand for water can be extreme, in some cases as much 400 units of water loss for every unit of CO<sub>2</sub> gained (Chapin III et al., 2002). An increased amount of atmospheric CO<sub>2</sub> allows for more efficient uptake of CO<sub>2</sub> and thus lower rates of stomatal water conductance at the leaf and individual organism levels (Curtis, 1996; Farnsworth et al., 1996; Urban, 2003; Herrick et al., 2004). However, just as it is both difficult and

unreliable to extrapolate changes in carbon uptake from the single tree to the forest, it is also difficult and unreliable to predict changes in transpiration at larger scales (Long, 1998).

## FOREST CARBON AND STAND DYNAMICS

### Stand development and carbon

Forest stands are dynamic components of the ecosystem in which carbon flux changes with size, age and species composition of trees. Although different species will influence stand development and carbon flux, general patterns exist for forest stands throughout the world (Oliver, 1992). Determining the developmental stage of a forest stand provides insight into the amount and nature of carbon flux, as different structural and age conditions influence photosynthetic rates and decomposition activity.

An important rule of thumb is that carbon allocation changes as tree size and stand structure increases. For example, seedlings allocate much of their carbon to shoot and root growth. One study found that paper birch (*Betula papyrifera*) and Douglas fir (*Pseudotsuga menziesii*) seedlings allocated 49% and 41% of absorbed isotopic carbon to their roots, respectively, and over 55% of this carbon was allocated to fine roots (Simard et al., 1997). Once seedlings have produced a sufficient root and foliar system, they are able to allocate carbon to stem height and then to diameter increment. At Hubbard Brook Experimental Forest in New Hampshire, the sapling stage of trees was found to contain the highest percent of dry weight biomass in stem or bole, and in another study, five-year-old loblolly pine (*Pinus taeda*) saplings were found to have allocated the majority of their carbon to stem growth (Whittaker et al., 1974; Retzlaff et al., 2001). As a tree matures, the percentage of total biomass held in stem and bole wood diminishes as the relative amount of biomass in woody branches increases (Whittaker et al., 1974). The amount of time it takes for a seedling to begin rapid height growth is dependent on stand species composition, temperature, light, soil and moisture conditions. For example, eastern hemlock (*Tsuga canadensis*) advance regeneration is able to survive without significant growth in the understory for decades until overstory conditions are right for it to continue height and diameter growth, in contrast to species such as eastern white pine (*Pinus strobus*) which will have high mortality at low light intensities (Burns and Honkala, 1990).

As saplings develop into poles and then mature trees, increasingly large quantities of carbon are stored in the stem. This process has been demonstrated by a study in which entire eastern white pine trees in Ontario, Canada were destructively sampled; researchers found that mature 65 year old trees contained 69% of their total biomass in their stem while only 25% of total tree biomass was in the stems of 2 year old trees (Peichl and Arain, 2007). Mature trees will eventually sequester less and less carbon as they become larger due to physical growth limitations such as water stress (Whittaker et al., 1974). Carbon is constantly lost over the life of a tree due to respiration and leaf, root, branch, and bark senescence; it may, however, be partially retained in the stand as coarse woody debris, leaf and branch litter, and soil organic matter.

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An important rule of thumb is that carbon allocation changes as tree size and stand structure increases.

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### Stand level carbon stocks

Stand level carbon stocks in the form of biomass and coarse woody material increase as a stand progresses through succession stages (Odum, 1969; Whittaker et al., 1974; Acker et al., 2000; Taylor et al., 2007). The rate of increase in biomass is not constant over the life of a stand (Song and Woodcock, 2003; Taylor et al., 2007); early stages of stand development have low rates of biomass accumulation due to trees re-establishing themselves on the site. A study of Siberian Scots pine (*Pinus sylvestris*) stands found that stand age had the largest influence on above ground net primary production (Wirth et al., 2002a). During stand initiation, net production (i.e., biomass accumulation) steadily increases and peaks during the stem exclusion stage (Odum, 1969; Whittaker et al., 1974; Acker et al., 2000). A study on a Douglas fir - and western hemlock (*Tsuga heterophylla*) - dominated stand in the Pacific Northwest of the United States found net primary productivity to be greatest during stem exclusion at 30 to 40 years (Song and Woodcock, 2003). Estimates of ponderosa pine (*Pinus ponderosa*) carbon uptake in newly developed stands was shown to increase exponentially as trees increased in size and recruitment of trees into the stand continued, with rates of increase ranging from 0.09 to 0.7 Mg of carbon per hectare per year depending on stand slope, aspect and soil conditions (Hicke et al., 2004). Stands with the same species composition growing on favorable sites will not only accumulate carbon at a higher maximum rate, but they will also reach this maximum rate sooner than stands on poor sites (Chen et al., 2002).

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While the magnitude of the carbon flux associated with mature stands is still being debated, it is important to consider that mature forest stands store far more carbon than early successional stands.

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Mature stands continue to accumulate carbon but at slower rates than stands going through the early stages of succession (Odum, 1969; Acker et al., 2000). Carbon storage in the living and dead biomass of a red spruce (*Picea rubens*) stand in Nova Scotia, Canada was found to follow a sigmoidal pattern across stand development, with 94 Mg of carbon per hectare in the youngest age class and 247 Mg of carbon per hectare in the 81-100 year age class, with lower amounts in the oldest age classes (Taylor et al., 2007). In the study on a Douglas fir- and western hemlock-dominated forest in the Pacific Northwest of the United States, a stand development model projected a gradual decrease in net primary production from 40 years until 300 years, when net primary production levels off (Song and Woodcock, 2003). The decreased rate of uptake is correlated with decreased woody biomass growth as stands age (Chen et al., 2002). Historically these old growth stands were considered neither sources nor sinks for atmospheric carbon. Although their rates of sequestration are lower, some may sequester far more carbon than previously thought; Carey et al., (2001) suggest that old growth forests in the Pacific Northwest are sequestering 145 Tg more carbon than terrestrial carbon models have predicted in the past for these forests. Similar results come from a model of a 200 year old eastern hemlock stand in central Massachusetts, which predicts that this forest has the ability to annually sequester more carbon in the living biomass with future climate change, because of higher atmospheric concentrations of CO<sub>2</sub>, than younger coniferous and deciduous stands had done in historical climates (Hadley and Schedlbauer, 2002).

While the magnitude of the carbon flux associated with mature stands is still being debated, it is important to consider that mature forest stands store far more carbon

than early successional stands (Thuille and Schulze, 2006). For example, a mature eastern white pine stand in southern Ontario, Canada held nearly double the carbon, both aboveground (100 tons per hectare) and below ground (56 tons per hectare), than a similar stand going through stem exclusion, which held 40 tons per hectare above ground and 39 tons per hectare below ground (Peichl and Arain, 2006).

### Stand disturbance effects

Natural and anthropogenic disturbances such as fire, disease, insect outbreaks, logging, and windthrow can alter the rate and/or direction of successional change and subsequently affect carbon flux in forest systems (Table 1).

**Table 1 Disturbance return intervals (in years) among different forest types**

DISTURBANCE	FOREST TYPE					
	Boreal	Temperate Hardwoods	Western N.A. Conifers	Tropical Rain Forests	Mediterranean Climate	Tropical Savanna
Fire	20-500	14-14,000	8-600	400-900	2-125	2-100
Insect Outbreaks	10-50	6-34	25-117			
Severe Wind Throw	50-75	150-1300	5-15	9-20		

Source: Fitzgerald, 1988; Huff, 1995; Lassig and Mocalov, 1998; Newbery, 1998; Walker, 1999; McKenzie et al., 2000; Thonicke et al., 2001; Lorimer and White, 2002; Ne'eman et al., 2002; Sinton and Jones, 2002; Burton et al., 2003; Ryerson et al., 2003; Felderhof and Gillieson, 2006; Fry and Stephens, 2006; Spetich and He, 2006; Shang et al., 2007; Bouchard et al., 2008

The severity and frequency of naturally occurring disturbances vary greatly within and between different forest types. The return interval for a forest is the approximate number of years between two disturbances. For major forest types, fire return intervals range from 2 to 14,000 years; insect outbreaks occur from 6 to 117 years; and wind throw is a perturbation that has a broad return interval ranging from 5 to 1,300 years (Table 1). The enormous variation in the size and type of disturbance and intervals between them within and across different forests, and the climates that drive them, is critical to understanding and managing stand dynamics and by implication carbon sequestration and storage.

Whenever forests are disturbed, they become sources of carbon as woody tissue dies, decomposes and releases stored carbon. The length of time it takes after a disturbance for a stand to become a carbon sink depends on the growth rate of newly established vegetation and the decomposition rate of downed woody material. When a forest is disturbed, some portion of available growing space is left unoccupied for a period of time while new and surviving vegetation grows to fill the site (Campbell et al., 2004; Humphreys et al., 2006). This lag time can range from months to centuries depending on disturbance type, climate, and site conditions. For example, 400 square meter logging gaps in a forest of the Bolivian Amazon were visually indistinguishable in aerial imagery from undisturbed forest after just three months (Broadbent et al., 2006) while boreal Scots pine stands may never reach

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The enormous variation in the size and type of disturbance and intervals between them within and across different forests, and the climates that drive them, is critical to understanding and managing stand dynamics and by implication carbon sequestration and storage.

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previous stocking levels after low intensity ground fires (Schulze et al., 1999; Wirth et al., 2002b).

Estimates of carbon flux within forests must therefore take into account disturbances in their various forms and frequencies (Cook et al., 2008). But, because each disturbance is unique, and species may respond to the same disturbance in different ways, determining the effects of a particular perturbation on stand level carbon budgets can be both difficult and imprecise. For example, in a study of a boreal forest fire in Canada by Randerson et al., (2006), analysis showed that when all the integrating effects of the fire (e.g. greenhouse gases, aerosols, carbon deposition on snow and sea ice, and post-fire changes in surface albedo) are accounted for, a decrease in radiative energy is expected when the fire cycle is over 80 years because surface albedo had a proportionately greater effect than fire-emitted greenhouse gases that only dramatically spiked radiation during the years immediately after the fire. This suggests that increases in boreal fires may not contribute to climate warming.

## **RESPONSE OF FORESTS TO INCREASED CARBON DIOXIDE**

### **Free Air Carbon Dioxide Enrichment (FACE) experiments**

Carbon dioxide enrichment studies provide insight into what the future may hold for the world's forests. Experiments are being conducted on a wide variety of terrestrial ecosystems in response to a predicted, continual increase in atmospheric carbon dioxide (IPCC et al., 2007). A review by McLeod and Long (1999) cited 145 references related to carbon dioxide enrichment experiments in multiple terrestrial ecosystems. These studies examined the response of ecosystem processes, including tree growth, to elevated levels of carbon dioxide in the ecosystem's local atmosphere by elevating ambient carbon dioxide to levels predicted for a specific year in the future. It is commonly believed that carbon dioxide enrichment will lead to an increase in vegetative growth in forest systems similar to that observed in carbon dioxide fertilized greenhouses. Such an increase would indicate that the growth of the stands being studied is currently limited by the concentration of atmospheric CO<sub>2</sub> (Millard et al., 2007).

There are two principal types of carbon dioxide enrichment experiments – free air carbon dioxide enrichment (FACE) and chamber carbon dioxide enrichment. FACE experiments elevate ambient levels by either releasing carbon dioxide gas into the air surrounding the study site or by releasing carbon enriched air into the study area (McLeod and Long, 1999). Other carbon dioxide enrichment experiments work with either fully enclosed chambers or open topped chambers which hold carbon enriched air on the site. FACE experiments are generally preferred over chambers when modeling ecosystem processes because they do not alter as many other environmental variables (Gielen and Ceulemans, 2001).

FACE experiments began in the 1980s with much of the research being done in agricultural systems (McLeod and Long, 1999). Forest ecosystems are still not well represented, primarily due to the difficulties and costs of creating and running

carbon dioxide enrichment towers in a forest. The annual cost of just the carbon dioxide necessary to operate a forest FACE experiment in the United States is over \$650,000, and represents one third of the annual budget of a site (DOE, 2002). Of the FACE experiments in forested ecosystems, only three are being conducted on sites larger than 5 hectares (Table 2). Only the Web FACE in Switzerland is being conducted in a forest stand that originated before 1980. The next oldest is the Duke Forest FACE which is in a 25 year old loblolly pine plantation (Asshoff et al., 2006; Keel et al., 2006; Oren, 2008). Globally, eight FACE experiments have been conducted in forested ecosystems (Table 2). There are three forest FACE experiments in the USA, three in Europe, one in Australia, and one in Japan. Of these, the Duke Forest FACE was the earliest; carbon dioxide enrichment began there in 1996 (Oren, 2008).

While informative, FACE studies are limited. There are large parts of the world and whole forest types in which no FACE studies have been conducted, notably Africa, mainland Asia, and South America. Of the eight studies, only one is in a tropical ecosystem (OZFACE), and none are in tropical moist forests or boreal forests. In addition to spatial gaps, FACE studies also lack structural diversity. Only the Web FACE is operating in a naturally regenerated forest (Asshoff et al., 2006; Keel et al., 2006), all others are being conducted in forest plantations, five sixths of which are younger than 20 years old. Many of these forests are not only young but also small, with some studies occupying less than 1 hectare. Although FACE studies provide us the best insight we currently have into ecosystem responses to elevated carbon dioxide, each of these shortcomings limits the scale and certainty of using results to predict ecosystem responses to carbon enrichment.

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Elevated carbon dioxide experiments have provided evidence that forest net primary productivity (NPP), and thus carbon uptake, increases when atmospheric carbon dioxide levels are increased.

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**Table 2 Global forest FACE experiments**

Name	Year CO2 enrichment began	Area (ha)	Ecosystem	Stand Initiation Year	CO2 Elevation	Reference
Duke Forest FACE Chapel Hill, NC, USA	1996	90	Planted <i>Pinus taeda</i> , Hardwood understory	1983	Ambient +200ppm	Oren, 2008
Oak Ridge FACE Tennessee, USA	1998	1.7	Planted <i>Liquidambar styraciflua</i>	1988	Ambient +200ppm	ORNL, 2003
Aspen FACE Rhinelander, WI, USA	1998	32	Planted <i>Populus tremuloides</i>	1997	Ambient +200ppm	
EuroFACE Viterbo Providence, Italy	1999	9	Planted <i>Populus</i>	1999 (coppiced in 2001)	550ppm	Pikkarainen et al., 2008
OZFACE Yabulu, QLD, Australia	2001	0.1	Planted tropical savanna	2001	460 and 550ppm	CSIRO, 2005
Web-FACE Hofstetten, Switzerland	2000	0.28	Temperate deciduous forest	circa 1900	600ppm	Asshoff et al., 2006
Hokkaido FACE Sapporo Japan	2003	0.014	Planted temperate deciduous forest	2003	500ppm	Eguchi et al., 2005
Bangor FACE Bangor, UK	2004	2.36	<i>pendula, Alnus glutinosa, fagus</i>	2004	Ambient +200ppm	Lukac, 2007

ppm= Parts Per Million, additional reference information from: [http://public.ornl.gov/face/global\\_face.shtml](http://public.ornl.gov/face/global_face.shtml)

Source: ORNL, 2003; CSIRO, 2005; Eguchi et al., 2005; Asshoff et al., 2006; Lukac, 2007; Oren, 2008; Pikkarainen and Karnosky, 2008.

### Results of Free Air Carbon Dioxide Enrichment (FACE) experiments

There is more room to explore the dynamics of forest carbon in relation to elevated atmospheric carbon dioxide, but what has been found so far is intriguing. Elevated carbon dioxide experiments have provided evidence that forest net primary productivity (NPP), and thus carbon uptake, increases when atmospheric carbon dioxide levels are increased. A study that analyzed the results from the Duke Forest FACE, Aspen Experiment, Oak Ridge, and EuroFACE experiments found that when atmospheric carbon dioxide levels were increased to a level predicted for the middle part of the 21<sup>st</sup> century, NPP increased by an average of 23(+/- 2)% (Norby et al., 2005). A follow-up study determined that nitrogen use increased on the three nitrogen limited sites (Duke Forest, Aspen Experiment, Oak Ridge) and nitrogen use efficiency increased on the EuroFACE site (Finzi et al., 2007). It is reasonable to attribute at least some of this increased nitrogen uptake to a reallocation of growth priority to root development (Chapin III et al., 1987; Norby and Iversen, 2006; Brunner and Godbold, 2007). These results raise questions about the long-term sustainability of NPP increases. As stands increase NPP, nitrogen may become progressively limiting and restrain future growth response (Finzi et al., 2006). The Duke Forest FACE – the longest running forest FACE program – has not yet shown such limitation, although the ecosystem level carbon-nitrogen ratio has increased (Finzi et al., 2006).

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As stands increase NPP, nitrogen may become progressively limiting and restrain future growth response.

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While FACE studies conducted in plantation forests in the stand initiation and stem exclusion phases showed significant increase in NPP, four year results in a mature temperate forest at the Web FACE showed that while the shoot length of some trees exposed to elevated carbon dioxide did increase, elevated carbon dioxide had no significant effect on stem growth (Asshoff et al., 2006). These results are difficult to extrapolate to larger scales and other mature stands as only 11 trees were exposed to elevated carbon dioxide, with 32 control trees.

A regeneration study at the Duke Forest FACE looked at the effect of elevated carbon dioxide on planted seedlings under low light conditions. Fourteen species of seedlings were planted, with a diverse light tolerance range between species. Only shade tolerant species were found to have better growth under elevated carbon dioxide and certain shade tolerant species were found to have higher survivorship (Mohan et al., 2007), a result that indicates that only those species not already limited by light were able to respond to carbon dioxide fertilization. Indeed, several studies have concluded that while it is possible that trees will increase use efficiency to overcome nutrient limitation driven by nutrient paucity (Ceulemans et al., 1999; Suter et al., 2002; Norby et al., 2005; Luo et al., 2006; Norby and Iversen, 2006; Springer and Thomas, 2007), light limitation appears to be insuperable (Teskey and Shrestha, 1985; Kerstiens, 2001; Urban, 2003). This will likely result in competitive advantages for shade tolerant species under elevated CO<sub>2</sub> (Hattenschwiler and Korner, 2000; Kerstiens, 2001; Mohan et al., 2007). These conclusions provide insight into potential future stand development patterns in forest systems; higher survivorship of shade tolerant regeneration may mean that total biomass for

individual stands will increase and/or the understory reinitiation stage of stand development could occur sooner.

FACE studies have helped elucidate the interactions between carbon and stand dynamics in forested ecosystems. Some interactions are far too complex to understand in just a few years, such as how carbon dioxide fertilization will interact with nitrogen limitation in future stands (Finzi et al., 2006; Millard et al., 2007; Iversen and Norby, 2008). It will take decades of studies to determine the true effects of increased atmospheric carbon dioxide on forest stand dynamics.

## PRECIPITATION AND TEMPERATURE AS OTHER CLIMATE EFFECTS

### Temperature change experiments in forest ecosystems

Over the next century, global temperatures are predicted to change at rates faster than at any time in historical records (IPCC et al., 2007). These rapid changes in temperature will alter future forest stand development and carbon cycling (Walther, 2004). Researchers have begun field experiments that simulate forests under predicted temperature changes in order to provide insight into the effects of climate change on these ecosystems (Ayres and Lombardero, 2000; Hanson et al., 2005; Danby and Hik, 2007; Hyvonen et al., 2007; Bronson et al., 2008; Lellei-Kovacs et al., 2008; Yin et al., 2008). Due to their ability to make large-scale predictions and the expense of on-the-ground experiments, there has been a heavy reliance on mathematically-based computer models (Plochl and Cramer, 1995; Sykes and Prentice, 1996; Iversen and Prasad, 1998; Beerling, 1999; Keller et al., 2000; Kirilenko et al., 2000; Bachelet et al., 2001; Schwartz et al., 2001; Dullinger et al., 2004; Iversen et al., 2004; Gibbard et al., 2005; Goldblum and Rigg, 2005; Hanson et al., 2005; He et al., 2005; Matala et al., 2006; Notaro et al., 2007; Xu et al., 2007; Delire et al., 2008; Leng et al., 2008).

The threat of warmer climates causes concern that higher temperatures will negatively affect species that have adapted to historical climate patterns, leading to shifts in species composition. While there is relative certainty that shifts in species' existing ranges will occur (Saxe et al., 2001; Walther, 2004; Wilmking et al., 2004; Hyvonen et al., 2007; Yin et al., 2008), predictions of which species will be affected and to what degree remain unreliable (Thuiller, 2004). One area where temperature-driven change will likely be dramatic is in boreal forests, where temperature is often a limiting factor, and species tolerant of low temperatures dominate the landscape (Hyvonen et al., 2007; Xu et al., 2007). Increased temperatures are likely to increase respiration in many species due to a longer growing season, and drought stress will occur in stands lacking the soil moisture needed to support the increased respiration (Saxe et al., 2001). Any drought stress may be moderated by plant reductions in stomatal conductance experienced at elevated CO<sub>2</sub> levels (Curtis, 1996; Heath, 1998; Herrick et al., 2004; Ainsworth and Long, 2005); however, the degree of response is highly species specific (Urban, 2003). Experiments in the boreal forest have shown reductions in growth induced by warmer temperatures and less relative moisture (Barber et al., 2000; D'Arrigo et al., 2004; Wilmking et al., 2004). A study of North

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It will take decades of studies to determine the true effects of increased atmospheric carbon dioxide on forest stand dynamics.

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American black spruce (*Picea mariana*), however, found no change in net ecosystem uptake of carbon dioxide despite a longer growing season, possibly due to increased respiration of the forest as a whole (Dunn et al., 2007). Where water is not a limiting resource, increased temperatures will likely allow for increased carbon uptake by trees; however, the total ecosystem response will be species and ecosystem specific (Boisvenue and Running, 2006; Matala et al., 2006).

### Precipitation fluctuation experiments in forest ecosystems

Water availability may be the most important factor driving NPP and consequently forest carbon dynamics (Tian et al., 1998; Del Grosso et al., 2008). As the Earth's climate continues to change, water availability in forest stands will change with temperature and precipitation. Precipitation regimes are predicted to change around the globe in relation to many factors including El Nino/Southern Oscillation (ENSO) events (Trenberth and Hoar, 1997). How these precipitation changes will affect carbon cycling and forest stand dynamics is unknown, but current drought studies provide some insight.

Seasonal changes in precipitation are likely to occur due to ENSO and other climatic events, thus creating seasonal droughts in some areas, such as the tropical forests in Borneo (Potts, 2003). Plant physiology tells us that if droughts occur in water-limited forests during the growing season, then carbon uptake by forests will decrease; if more rainfall occurs during the growing season in areas where water is limiting, we might expect more uptake of carbon by forest stands (Larcher, 2003). This effect has been observed in a Scots pine forest in the Rhine plain, where a relatively cool/moist growing season led to a near doubling in the carbon sink as compared to a relatively warm/dry year (Holst et al., 2008). Care should be taken when making broad generalizations on the degree to which growth and subsequently carbon uptake of forest trees is affected by changes in precipitation because these responses are highly species- and site-specific and wider trends remain unclear. What is clear, however, is that soil moisture plays an important role in controlling carbon storage in forests (Tian et al., 1998).

A precipitation study done in a temperate forest ecosystem in the Appalachian mountains (USA) found that forest growth in wet years was as much as 3 times greater than growth in dry years (Hanson et al., 2001). This same study found that spring droughts reduced growth to a greater degree than droughts later in the growing season, with greater mortality in saplings as compared to mature trees. This study suggests that the timing of droughts will play a major role in controlling future stand development. If droughts occur during dormant seasons, the effects of precipitation regime changes may be minimal or only expressed in long-term soil drying. Also noteworthy from the Hanson et al. (2001) study was the fact that mature trees were less affected by drought than were understory saplings, demonstrating the resilience of current forest stands to climatic changes. This is consistent with established ideas about the relative sensitivity of regenerating stands (Finegan, 1984). Mature Scots pine stands in Siberia were found to have a positive correlation between

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above ground net primary productivity and growing season precipitation (Wirth et al., 2002a). These results suggest that current stands will endure climatic changes in part through changes in species composition, which is partially as a result of changes in moisture conditions (Hanson et al., 2001; Thuiller, 2004; Frey et al., 2007).

### **Combined effects of climate change on forest ecosystems**

Experiments investigating the combined effects of climate change – increased carbon dioxide, temperature changes, and precipitation changes – are more realistic than those exploring any single factor (Hyvonen et al., 2007). Despite this, single factor studies are far more common than those exploring multiple climate variables. The true dynamics of these systems are unknown, and models predicting carbon cycling using multiple variables are often very sensitive to changes in site, species, and productivity of forests (Hanson et al., 2005). Results from a model representing dynamics in an upland oak forest in the eastern United States demonstrate the sensitivity of model results to external factors affecting forest stand dynamics, such as nutrient availability, temperature and water availability (Hanson et al., 2005). The model combined the effects of increased atmospheric carbon dioxide, temperature, precipitation and ground level ozone. Results showed that without any other changes the forest would reduce its net exchange of carbon dioxide by 29%, therefore increasing sequestration. However, when physiological adjustments (such as longer growing seasons) were incorporated into the model, results showed net exchange of carbon dioxide increasing by 20%, and therefore *releasing* of carbon. While helpful, the results of stand development models are based on a series of assumptions, and will vary widely as those assumptions are revised. An example might be the variability in different possible combinations of temperature flux across seasons. Increases or decreases in temperature (summer temperatures may increase, winter ones may stay the same or vice-versa) will have repercussions on phenology, herbivory, snow melt, and many other interacting biological and physical factors that make predictions so hard to make regarding the ultimate effects on carbon flux and storage in ecosystems.

The complexity of forest stand dynamics makes controlling variables in manipulation experiments extremely challenging and expensive and for that reason scientists rely heavily on multifactor models (Luo et al., 2008); however, the majority of climate change models for forests account for just 1 to 5 variables (Curtis et al., 1995; BassiriRad et al., 2003; Hanson et al., 2005; Bandeff et al., 2006). Long-term studies are needed to investigate the interactions of changes in carbon dioxide, temperature, and precipitation, as these will provide the best data for making predictions about carbon cycling in future forest stands (Karnosky, 2003).

### **CONCLUSIONS AND SUMMARY RECOMMENDATIONS**

Carbon cycling in forests is a complex process with many variables. General patterns of stand carbon cycling are universal, but the temporal dynamics of these patterns are very site specific. We suggest that the following findings are important to consider:

- Stands accumulate carbon as they progress through succession. Most studies show that the greatest rate of carbon uptake occurs during the stem exclusion stage, but even mature stands sequester and store large quantities of carbon. Recent studies have shown that this can be consequential – even for old growth – particularly when such old forests represent significant portions of large areas such as the Amazon and Congo basins (Lewis et al., 2009).
- Disturbances to forest ecosystems cause a release of carbon as woody vegetation decomposes. Future climatic conditions will play a major role in carbon cycling in forest stands, and conversely, future stand conditions will influence climate.
- FACE experiments provide evidence that with increased atmospheric carbon dioxide some species will have increased growth; however, further research is clearly needed.
- Future precipitation patterns and moisture regimes will shape forest structure, species composition and productivity, but those changes will vary greatly with both site and timing.
- The combined effects of climate change are being investigated, but often there are too few variables being considered, making the global application of results from these studies somewhat questionable.

Areas of uncertainty in forest carbon science at the stand level provide numerous opportunities for future research. A major area of uncertainty in current research is the long-term effect of changing climates on forest ecosystems. The majority of FACE studies (Table 2) and drought studies are less than 20 years old. Further investigation of below ground carbon dynamics in forest systems is also needed (Ceulemans et al., 1999; Curtis et al., 2002). There are entire forest types with little research related to carbon cycling at the stand level, such as the tropics (Clark, 2007; Stork et al., 2007), and the scale of the existing studies also leaves much to be investigated. The largest FACE study currently operating is the Duke Forest FACE on 90 hectares; the second largest is Aspen FACE at 32 hectares (Table 2). With only two large-scale (greater than 30 hectares) FACE studies in forest ecosystems, extrapolating the effects of increasing concentrations of atmospheric carbon dioxide to global scales is untenable. What we know about forest stand carbon cycling provides a quality base for future experiments, but leaves much to be desired in predicting forest carbon budgets (Karnosky, 2003).

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## Chapter 4

# Carbon Dynamics of Tropical Forests

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### EXECUTIVE SUMMARY

Tropical forests, a critical resource affecting world climate, are very diverse, largely because of variations in regional climate and soil. For purposes of this analysis they have been divided in four broad forest types – ever-wet, semi-evergreen, dry deciduous, and montane. Existing literature on climate and tropical forests suggests that, compared to temperate and boreal forest biomes, tropical forests play a disproportionate role in contributing to emissions that both affect and mitigate climate. This chapter describes the geographical extent of tropical forests and their role in terrestrial carbon storage, uptake (through processes of photosynthesis), and loss (through plant respiration and microbial decomposition of dead biomass). A review is provided of current knowledge about the role of disturbance (natural and human caused) in affecting the carbon balance of tropical forests, considering the impacts of windthrow, fire, drought, and herbivory. The chapter concludes with an analysis of the threats to tropical forests and how they may influence climate change and elevated CO<sub>2</sub>. Findings of this review are summarized in the section below under “what we know” and “what we don’t know” about the carbon dynamics of tropical forests.

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### What do we know about carbon storage and flux in tropical forests?

- Tropical forests contribute nearly half of the total terrestrial gross primary productivity. About 8% of the total atmospheric carbon dioxide cycles through these forests annually.
- Tropical forests contain about 40% of the stored carbon in the terrestrial biosphere (estimated at 428 Gt of carbon), with vegetation accounting for 58% and soil 41%. This ratio of vegetation carbon to soil carbon varies greatly by tropical forest type.



- As tropical ever-wet forest soils become drier, litter decomposition and release of CO<sub>2</sub> from soil may slow. However, studies show that release of methane, which has a higher global warming potential than CO<sub>2</sub>, increases as soils dry. The cause of the methane increase is suspected to be related to increased termite activity.
- Tropical ever-wet and semi-evergreen forests in the Amazon and southeastern Asia typically suffer from droughts during ENSO events (El Niño – La Niña). In the short-term, tropical forests may be resilient to drought. However, this may be offset by increased vulnerability to fire after both short- and long-term droughts. These droughts are more severe during strong El Niño years. In tropical ever-wet forests, where droughts are rare, mortality may increase during strong El Niño years due to severe drought, while seasonal semi-evergreen forests may experience relatively little change.
- Old growth ever-wet and semi-evergreen forests are experiencing accelerated stand dynamics and increasing biomass. Studies have shown that there has been a net increase in biomass in recent decades in Amazonian and Central African forests that is potentially a significant response to increased atmospheric CO<sub>2</sub>.
- Expanding crop and pasture lands have a profound effect on the global carbon cycle as tropical forests typically store 20-100 times more carbon per unit area than the agriculture that replaces them. The use of fire to clear forested lands may exacerbate changes to carbon cycling since fire fills the atmosphere with aerosols, thereby reducing transpiration.
- There are proportionately higher amounts of fine root biomass (as compared to other vegetative parts – e.g. leaves, stem) in infertile soils as compared to fertile soils. Infertile soils (e.g. oxisols) make up a greater proportion of the African and South American upland ever-wet and semi-evergreen forest than any other soil type.
- CO<sub>2</sub> production in tropical soils is positively correlated with both temperature and soil moisture, suggesting that tropical rain forest oxisols are very sensitive to carbon loss with land use change.

#### **What don't we know about carbon storage and flux in tropical forests?**

- Uncertainties in both the estimates of biomass and rates of deforestation contribute to a wide range of estimates of carbon emissions in the tropics. More studies are needed.
- In response to elevated CO<sub>2</sub>, many models predict increased forest productivity, but recent studies suggest that stem growth rate actually decreased in the last twenty years largely due to increased nighttime temperature, decreased total precipitation, and increased cloudiness.

- Direct measurement of below-ground carbon stored in roots is often very difficult even with the most thorough root collection. Current estimates of root soil carbon in tropical forests could be underestimated by as much as 60%. Contrary to past assumptions, a significant portion of stored carbon exists below ground in tropical forests.
- Since many climate models predict further soil drying and increased litter fall in tropical forests, understanding the role of soil microbial communities in processes within the litter layer, belowground biomass, and soil carbon is key.
- Only three studies have analyzed land surface-atmosphere interactions in tropical forest ecosystems. It is essential to understand how carbon is taken up by plants and the pathways of carbon release, and how increasing temperatures could affect these processes and the balance between them.
- Better estimates are needed of the amount of mature, secondary, and disturbed forests in the tropics in order to better predict changes in carbon storage trends and the threat of release of this terrestrial sink.
- The effects of elevated atmospheric CO<sub>2</sub> and global climate change on herbivory and other plant/animal interactions in tropical forests are not well understood. Little research has been done in this area.
- Tropical dry deciduous and montane forests are almost a complete unknown because so little research has been done on these forest types. While the majority of dry deciduous forests in the Americas and Asia have been cleared, there is still a significant amount remaining in Africa.

#### **What are the major influences on carbon storage and flux in tropical forests?**

- First and foremost, the primary risk to the carbon stored in tropical forests is deforestation, particularly converting forests to agriculture. Current estimates of carbon emissions from tropical deforestation vary greatly and are difficult to compare due to differences in data sources, assumptions, and methods. Developing and incorporating multiple variables into new and existing ecosystem models for tropical forests is essential to determining carbon fluxes and future effects of deforestation and climate change.
- Combined climate-carbon cycle models predict that tropical forests are vulnerable to both short- and long-term droughts. The effects of drought will vary, depending on the forest type, whether or not the forest is water-limited, and the counter-effects of increased sunlight. At least in the short-term, tropical forests may be resilient to drought. However, this may be offset by increased vulnerability to fire after both short- and long-term droughts.
- Changes in soil moisture affect not only the response of plant species and communities, but also the population dynamics of animals, fungi and

microbes, which in turn will have impacts on herbivory and decomposition. Elevated CO<sub>2</sub> reduces nitrogen-based defenses (e.g., alkaloids) and causes an increase in carbon-based defenses (e.g., tannins). As leaves exhibit lower nutritional value, herbivory may increase substantially to compensate.

- All large-scale wind and rain events are episodic and occur at relatively long time intervals that are difficult to predict. However, they drive the successional dynamic of forests, and therefore by implication, the above- and below-ground carbon stocks. Little to no work has been done on assessing and including this dynamism in the development of regional carbon models predicting future change. The assumption is that small-scale disturbances in old-growth forests will remain the dominant phase of growth.

### **How might the carbon status of tropical forests change with changing climate?**

- The difference between the annual stand level growth (uptake: 2%) and mortality (release: 1.6%) of Amazonia is currently estimated to be 0.4%, which is just about enough carbon sequestered to compensate for the carbon emissions of deforestation in the region. This means that either a small decrease in growth or a small increase in mortality in mature forests could convert Amazonia from a sink to a source of carbon.
- It is difficult to model carbon flux and productivity in tropical forests due to their structural and age complexity and species composition. As a result, few ecosystem process models have been developed, parameterized, and applied within tropical forest systems. Nevertheless, it is a reasonable assumption that rising temperatures will increase the rate of most if not all biochemical processes in tropical plants and soils.
- In response to elevated CO<sub>2</sub>, many models predict increased productivity, both in semi-evergreen forests of the Amazon and central Africa. However, on ever-wet sites in Panama and Malaysia, stem growth rate actually decreased from 1981-2005 largely due to increased nighttime temperature, decreased total precipitation, and increased cloudiness.
- Old-growth tropical forests are experiencing accelerated stand dynamics and increasing biomass. Most climate models and forest carbon balance models do not take forest composition into account. Forests with accelerated or “faster” dynamic have less biomass due in part to ecophysiological differences in plant growth.
- A warmer climate could drive low elevation forests to higher elevations or extend the range of tropical seasonal forests. However, if there is more deforestation in these seasonal and dry areas, there may be fewer species available that can migrate and adapt to warmer climates with drier soils.
- Many future climate scenarios predict soil drying in Amazonia and a general reduced capacity of the ecosystem to take up carbon. Understanding how

tropical forests respond to water stress could be important because canopy-to-air vapor deficits and stomatal feedback effects could determine how tropical forest photosynthesis responds to future climate change.

- As tropical forest soils become drier, litter decomposition and its release of CO<sub>2</sub> from soil may slow in response to less water availability. However, there is also some evidence that methane release may increase as soils dry out.
- If drought becomes more common in tropical ever-wet and semi-evergreen forests, as some climate models predict, the likelihood of human-induced fires escaping and impacting large portions of the landscape increases.

**Keywords:** Amazon, Borneo, Central Africa, disturbance, drought, dry deciduous, El-Nino, ever-wet, fire, insects, oxisols, semi-evergreen, ultisols

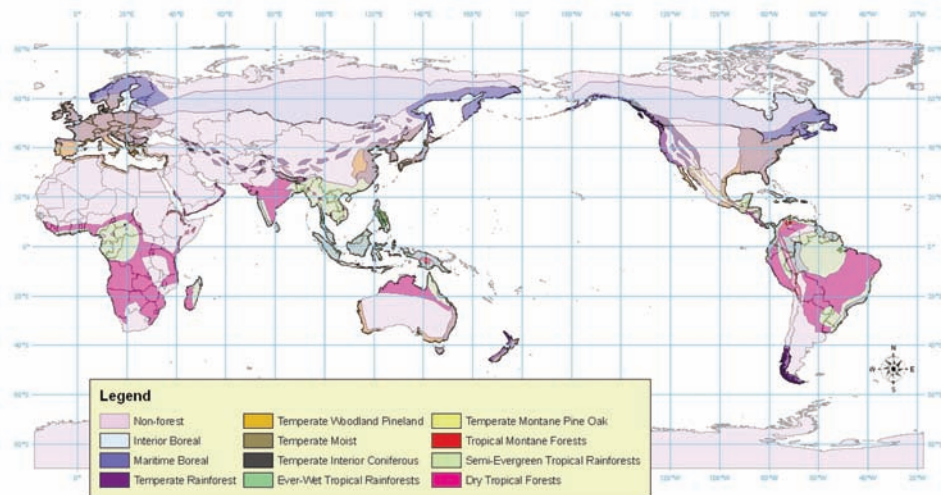
## INTRODUCTION

This chapter reviews current literature about carbon cycling in tropical forests. It first broadly describes the different kinds of tropical forests, where they are found, their current and past extent, and their role in terrestrial carbon storage. Secondly it describes how and where carbon is allocated in tropical forests, how carbon cycles, and how climate change could affect this cycling. Thirdly, we discuss how changes in carbon storage may occur through uptake, via photosynthesis, and through loss, via respiration and decomposition. Next is the role of disturbance and its potential effects on stored carbon. Finally, the chapter concludes with a review of some of the threats to tropical forests and how they may influence climate change and elevated CO<sub>2</sub>.

The level of interest in tropical forests has increased in recent decades due to global issues of climate change, biodiversity loss, and land use change (predominantly conversion of forest to agriculture). Globally, the tropical rain forest regions of Southeast Asia, South America, and Central Africa are some of the most rapidly developing areas of the world in terms of population growth, land conversion, and urbanization (Houghton, 1991a; Soepadmo, 1993; Nightingale et al., 2004). Tropical deforestation is one of the main contributing factors to the increase of CO<sub>2</sub> in the atmosphere (Houghton, 1991a; Houghton, 1991b). Despite their importance and impact on the global carbon cycle, there is a lack of systematic assessment, and therefore knowledge, about the carbon pools and fluxes in tropical forests (Dixon et al., 1994; Lal and Kimble, 2000; Nightingale et al., 2004). Although some generalizations can be made about tropical forest biomes across the globe, such highly diverse, complex systems warrant closer attention in order to make better estimates and predictions of global carbon budgets. Moreover, there is a tendency in carbon-related policy making to overlook the carbon cycle's interconnectedness with other biogeochemical cycles, such as water and nitrogen. None of these cycles occur in isolation; it is important to remember that carbon is related to biodiversity, water storage and filtration, and other ecosystem values.

Tropical forests occupy a broad range between the Tropic of Cancer and the Tropic of Capricorn, where moist air rising from the equatorial region loses this moisture in the form of precipitation as it descends over the tropics and subtropics (Heinsohn and Kabel, 1999). These forests cover approximately 12% of the land surface and account for 50% of global forest area (Figure 1). Approximately 8% of total atmospheric carbon dioxide cycles through these regions annually (Malhi et al., 1998). Tropical forests are responsible for nearly half of the total terrestrial gross primary productivity (Malhi et al., 1998). They therefore play a major, yet poorly understood, role in the cycling of carbon (Frangi and Lugo, 1985; Soepadmo, 1993; Foody et al., 1996; Malhi et al., 1998).

**Figure 1** Original extent of boreal, temperate, and tropical forest types of the world prior to land clearing



## TROPICAL FOREST SYSTEMS

Tropical forests can be divided into four broad types: i) ever-wet (often called rainforest); ii) semi-evergreen; iii) dry deciduous; and iv) montane (Figure 1). Forests types have been categorized in relation to both the amount of precipitation and degree of seasonality as the main driver of productivity and decomposition, and hence carbon sequestration and loss.

### Forest type descriptions

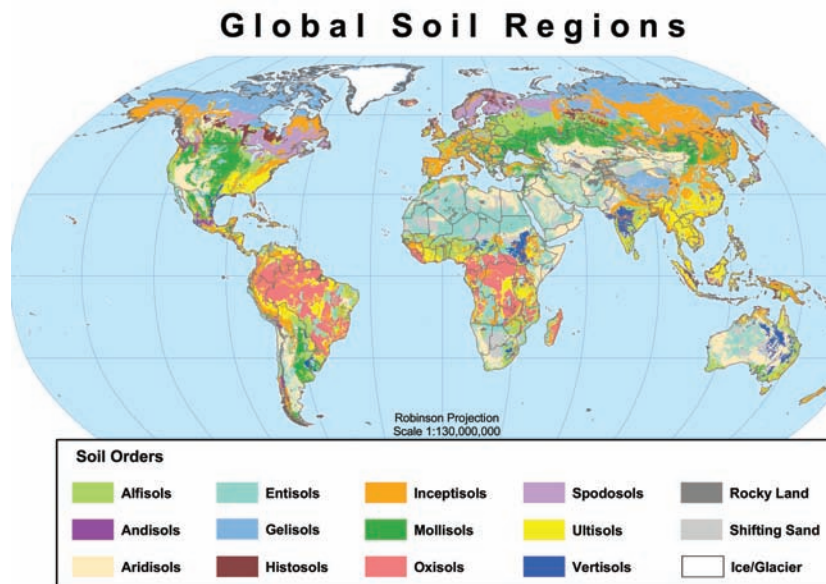
#### *Ever-wet forests*

Tropical ever-wet forests receive at least 100 mm of precipitation each month and at least 2,000 mm per year (Ricklefs, 2001). Vegetation tends to be dense and of several strata (e.g., canopy emergents, canopy, lianas, epiphytes, treelets, shrubs, and herbs).

High temperatures coupled with high moisture lead to rapid decomposition in these systems. The highly productive vegetation has adapted to this climate with the ability to immediately take up nutrients. As a result, many of the nutrients of tropical rain forest ecosystems are contained within the vegetation. Poorly planned and intensive logging or land clearance and burning can result in the loss of nutrients and render the landscape unproductive (Ricklefs, 2001; Vandermeer and Perfecto, 2005).

The majority of soils in ever-wet forests tend to be well-weathered ultisols, which are acidic, vary in fertility depending upon underlying geology, have relatively high cation exchange capacity, and are very susceptible to erosion. However, this is by no means consistent across the biome. Inceptisols predominate on young foothills, and andisols dominate on volcanic substrates. Both are characteristic of Central America and volcanic islands such as Sumatra, and both are fertile but strongly erodable (Figure 2).

Figure 2 A map depicting the major soil orders of the world



Source: USDA, Natural Resources Conservation Science, Soil Survey Division, *World Soil Resources*, 2005.

In West Africa, the ever-wet forest occurs along a thin strip of coast from Liberia to Ghana. It starts again in southeastern Nigeria, expanding across Cameroon and around the Gulf of Guinea. The wettest area of the region is the Cameroon Highlands, where rain fall at the base of Mt. Cameroon can reach over 12,000 mm per year. However, most of the area would be classified as marginally ever-wet, with rainfall in most of the range barely over 2,000 mm. Because of its ease of access for human populations, most of the coastal forest that historically spanned Cote d'Ivoire, Ghana, Nigeria, and Cameroon has been lost during the periods of French and British

colonization with the commercialization of plantation crops such as coffee and cocoa (1930-1960), and now oil palm. Forests in these countries are now largely reduced to small degraded patches.

The ever-wet rainforests were once expansive, covering all of eastern Central America (Atlantic Coast) from northern Costa Rica south through Panama, and along the Pacific coastal mountains of Columbia and northern Ecuador (Figure 1). The other wet evergreen forest of the Americas covers the eastern foothills of the Andes and forms the upper basin of the Amazon. The wettest forests in Latin America are those straddling the Andes in the region known as the Chocó on the Pacific coast range of Colombia, and the upper Amazon of Ecuador. The Atlantic region of Central America has been difficult for people to access and still remains extensively forested, particularly in Panama, as well as the upper Amazon regions bordering Colombia, Ecuador, and Peru.

The core Asian ever-wet forest can be considered the most moderated in seasonality largely because the land-sea margin and north-south mountain ranges serve as important sources of convectional and orographic precipitation during inter-monsoonal wet seasons. The heart of the ever-wet rain forest is in Borneo, Sumatra, New Guinea, and the Malay Peninsula, an area that makes up the largest extent of ever-wet rainforest in the world. Small areas also exist in southwest Sri Lanka, parts of the Western Ghats of India, and Mindanao in the Philippines.

Asia has had the longest legacy of rainforest commercialization (dating back 2,000 years), largely through maritime trade between Indian, Arab, and Chinese traders and the regional peoples. India's and Sri Lanka's forests are now largely restricted to the mountains and uplands of the countries, where historical land use for intensive rice cultivation, private tree garden systems, and plantation agriculture (tea, rubber, coconut – 1850-1950) has been happening much longer than elsewhere in Asia. Most of the ever-wet forest in the Philippines and Thailand is now confined to degraded patches, first logged over, and then subsequently and incrementally converted to village agricultural projects, many of which subsequently failed and are now wastelands (1940-1985). The Malay Peninsula had most of its lowland forest converted to plantation crops (rubber and oil palm) starting with the British (1900) but accelerating post independence (1948) such that most of the lowlands had been converted by 1980.

Substantial forest remains in the highlands but it is heavily cut over. A similar story exists for Sarawak and Sabah, the two east Malay states on Borneo. However, for these states, land conversion of the lowland forests occurred very rapidly and recently (1970-2000). Indonesia embarked on rapid logging and land conversion of its wet evergreen forests in Kalimantan (Indonesian Borneo) and Sumatra initially for colonization schemes (1970-1980), then more substantively as logging concessions. Subsequently, much of the logged over forest has been converted to oil palm plantations. In Borneo and Sumatra, both countries (Malaysia and Indonesia) have now embarked on clearing the remaining logged over forest for *Acacia mangium* pulp plantations or for oil palm (1995-ongoing). The remaining forested areas are restricted to the most unproductive soils and upland regions that are difficult to

access. New Guinea (Papua and Irian Jaya, Indonesia) can be considered the last frontier of remaining large intact forest within the region, although much of it has been allocated for logging concessions (1990-ongoing).

### *Semi-evergreen forests*

Tropical seasonally moist forests, also known as tropical semi-evergreen, like ever-wet forests, receive greater than 2,000 mm per year of rain. However, the forest type is more strongly seasonal (in Asia – monsoonal) with extended dry periods and then high periods of rain. Wet periods are generally longer than dry periods.

Soils are usually oxisols (or sometimes spodosols). They are both infertile soils, and acidic. Oxisols are highly weathered, with high clay content, and low cation exchange capacity (Clark et al., 1999; Vitousek and Sanford, 1986). Oxisols dominate the uplands of the core Amazon and Congo basins. Alfisols, which are relatively more fertile, are usually found in seasonally drier climates that are not so strongly monsoonal. They predominate particularly in Indochina (India, Burma, Thailand, Cambodia, and Vietnam) (USDA, 2002; Vitousek and Sanford, 1986) (Figure 2).

The greatest extent of semi-evergreen forest was that of the central and lower Amazon basin and the upper Orinoco of southern Venezuela. Much of the forest in the heart of the Amazon remains largely intact, but has been logged over through the use of the extensive river systems. Coastal and floodplain forests of the major rivers that flow into the Amazon have largely been converted to agriculture. The outer periphery of the basin (particularly on the southern side) and the coastal Atlantic forest of Brazil has retreated considerably because of colonization schemes and large land conversion to commercial soybean and ranching (1970-ongoing).

In West Africa, semi-evergreen forest dominates behind the band of coastal ever-wet rainforest, and can be considered a transition zone to dry deciduous forest further inland. Semi-evergreen forests also predominate in the central Congo River basin. The forests are generally more seasonal than those of the Amazon, with greater levels of deciduousness exhibited by some canopy species. Because of the difficulty of access, the inner core region of the Congo (Central African Republic, Republic of Congo, Democratic Republic of Congo, Gabon) largely remains whole, though current timber extraction is high (1990-ongoing). Both Amazon and Congo semi-evergreen forest can be considered by far the most important and largest tracts of tropical forest left in relation to forest carbon and climate change.

Indochina is the third region with semi-evergreen forest. The forest is found in parts of the Philippines, southern Thailand, northeast India/Burma, southeast Cambodia, southern Vietnam, northeast Sri Lanka, and the Western Ghats of India. The forest type is highly fragmented because of the physical geography and climate. This is an area of high soil fertility so most of the forest has been cut down and converted to agricultural use. The remaining forest patches are mostly degraded.

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Both Amazon and Congo semi-evergreen forest can be considered by far the most important and largest tracts of tropical forest left in relation to forest carbon and climate change.

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### *Tropical dry forests*

Tropical dry deciduous forests can be defined as those forests which shed their leaves during a dry season due to low water availability (Ricklefs, 2001). They are located in the tropics and subtropics, mainly in Latin America, Africa, India, Australia and parts of Southeast Asia (Bullock et al., 1995). They can be located in rain shadows of mountainous regions and near mid-latitudes of convergence. Longer and more severe dry seasons support tropical dry seasonal forests and savannah ecosystems (Ricklefs, 2001). Soils tend to be alfisols, entisols, and inceptisols (USDA, 2002).

This gradient in moisture regimes across the biome has led to much debate over the extent of dry deciduous forests vs. savannahs in the drier tropics (Bullock et al., 1995). Dry deciduous forests are found on more fertile sites than savannas, although they can occur in the same climate zone. In many places, human intervention and fire govern the line between forest and savannah (Bullock et al., 1995). Tropical dry forests receive far less attention than tropical ever-wet and semi-evergreen forests, even though conservation concerns are high due to increased land use conversion, habitat fragmentation, and high levels of biodiversity and structural diversity (Bullock et al., 1995).

In Central America dry deciduous forest used to dominate the Pacific side of Nicaragua, Costa Rica, and Panama. Most of this forest has been cleared for ranching, but some is now coming back as secondary forest because ranching cannot be sustained due to soil degradation. Dry deciduous forest still dominates much of the Yucatan (Mexico, Guatemala). In South America, dry deciduous forests were extensive across the coast range and interior Pacific sides of the Andes in Colombia and Ecuador and in the Caribbean coastal mountains and interiors of the lower Orinoco. Most of this forest has now been converted to ranch lands, although in places second growth is coming back. In the southern rim of the Amazon basin in Brazil, Bolivia, and northern Argentina, dry deciduous forests have been cleared for plantation agriculture and ranching. Little exists today except for some remnant patches.

Africa has the largest dry deciduous forest remaining, making up the miombo woodlands of Malawi, Zimbabwe, Tanzania, Angola, southern Democratic Republic of Congo, Mozambique and Botswana. It is an important resource for local people for firewood, timber, and grazing, and in some areas is heavily deforested. Nevertheless, the woodland in many areas remains relatively intact.

Dry deciduous forests also exist as small residual patches in what was extensive woodland in south India (east of the Ghats) across central India, Central Burma and Thailand, and interior Cambodia. Most is now converted to small-holder farms and degraded forest patches. Australia has considerable dry eucalypt woodland remaining across West and South Australia and in the north (Queensland and Northern Territories). However, a still greater portion has been cleared for raising sheep and for commercial agriculture.

### *Montane tropical forests*

Montane tropical forest is the smallest in area (current and historical) compared to the other tropical forest types. Montane forest occurs above 3,000 m above sea level

and is characterized by high precipitation (> 2,000 mm per year) and lower amounts of radiation because of cloud cover. Epiphytes, particularly bromeliads, often characterize the groundstory and canopy. The greatest amount of forest of this type is in Latin America down the Cordillera of Central America and along the northern Andes from Venezuela to Peru. Asia has montane forests that are numerous but small, being largely confined to the tops of the Western Ghats (India), the central range of Sri Lanka, the highlands of Thailand, Cambodia, and Vietnam and the Ginteng Highlands of Peninsula Malaysia. Larger extents of montane forest exist as the backbone of the islands of Borneo and Sumatra, and the volcanoes of the Philippines. The greatest extent is on the plateaus and the jagged mountains of Papua New Guinea. Africa has only small amounts of montane forest on the slopes of the inland mountain systems of Central (Rwanda, Burundi) and East Africa (Kenya, Uganda and Tanzania).

The soils of montane forests are often some of the most productive and would be mostly classified as inceptisols, which are high in soil organic matter (soil carbon), but are erosion-prone because of steep slopes. Many of the mountain regions adjacent to cities (Kuala Lumpur, Malaysia; Colombo, Sri Lanka; David, Panama; Quito, Ecuador; Bogota, Colombia; Nairobi, Kenya) have had their forests cleared for vegetable production, tea and coffee cultivation, and dairy. Much of the organic matter is lost through decomposition, and once depleted, such areas often revert to fire-prone invasive grass and fern lands.

### **POOLS OF CARBON IN TROPICAL FORESTS**

Tropical forests contain about 40% of the carbon in the terrestrial biosphere, an estimated 428 Gt of carbon, with vegetation accounting for 58%, soil 41%, and litter 1% (Soepadmo, 1993; Watson et al., 2000). The carbon budget across tropical forest types can be further broken down into interrelated components: aboveground biomass, belowground biomass, litter, and soil carbon (Table 1). Aboveground biomass consists of live stems and large branches and often includes coarse woody debris (Malhi et al., 2004). Belowground biomass includes all root mass (Robinson, 2007). Litter usually includes twigs, leaves, reproductive parts and other small biotic debris with short residence times (Malhi et al., 2004). What is included in soil carbon measurements, and how it is allocated within these categories, can vary from study to study. For example, some studies include the litter layer with the soil carbon analysis (e.g., Schwendenmann and Veldcamp, 2005). Other researchers separate roots, large organic debris, and rocks from soil for analysis (e.g., Cleveland et al., 2007). No one method is superior. Each method comes with its own advantages and disadvantages depending on the research question. While the use of categories helps to facilitate measurement and analysis, it is also necessary to understand the level of flux between the various carbon pools. This is important not only to correctly measure each component of the carbon cycle, but also to determine the strengths of the links between pools and other biogeochemical cycles.

Table 1 A summary of carbon studies in tropical forests

Source	SITE CHARACTERISTICS			CARBON POOLS				
	Location	Forest type	Age	Aboveground biomass	Coarse/fine woody debris stock	Coarse woody debris annual	Belowground biomass input	Soil
Malhi et al., 2004	Lowland Amazon	Ever-wet/semi-evergreen	Mature/old growth	increment of 1.5-5.5 Mg C ha <sup>-1</sup> a <sup>-1</sup>				
Baker et al., 2007	Upper Amazon/Peru	Ever-wet	Mature/old growth		24.4 +/- 5.5 Mg C ha <sup>-1</sup>	3.8 +/- 0.2 Mg C Ha <sup>-1</sup> y <sup>-1</sup> with a 4.7 +/- 2.6 y <sup>-1</sup> turnover		
Clark et al., 2001	Costa Rica	Ever-wet	Mature/old growth	Increment of 1.7-11.8 Mg C ha Y <sup>-1</sup> (lower bounds); 3.1-21.7 Mg C Ha <sup>-1</sup> Y <sup>-1</sup> (upper bounds)				
Clark et al., 2002	Costa Rica	Ever-wet	Mature/old growth		fallen - 22.3 Mg C Ha <sup>-1</sup> ; standing - 3.1 Mg C Ha <sup>-1</sup>	2.4 Mg C Ha <sup>-1</sup> with a 9 y turnover		
Nepstad et al., 1994	Lowland Amazon	Semi-evergreen	Mature/old growth					Between 1-8 m soil depth more soil carbon than above ground biomass; 15% turnover
Feeley et al., 2007	Lowland Malaya; Panama	Ever-wet	Mature/old growth	Decreases in growth recorded in 24-71% trees in Panama; 58-95% trees in Malaya				
Espeleta and Clark, 2007	Costa Rica	Ever-wet	Mature/old growth				10 fold variation over 7 y period; 4 fold change across edaphic gradient of soil water availability/fertility	
Houghton et al., 2001	Amazon	Ever-wet/Semi-evergreen	Mature/old growth	Mean total standing and below ground biomass 177 Mg C Ha <sup>-1</sup>				
Lewis et al., 2004	Amazon	Ever-wet/Semi-evergreen	Mature/old growth	Basal area has been increasing at 0.10 +/- 0.04 M2 Ha <sup>-1</sup> Y <sup>-1</sup> between 1971-2002				
Lewis et al., 2009	Central Africa	Ever-wet/Semi-evergreen	Mature/old growth	Above-ground biomass has been increasing at 0.63 Mg C Ha <sup>-1</sup> Y <sup>-1</sup>				
Phillips et al., 2008	Amazon	Ever-wet/Semi-evergreen	Mature/old growth	Above-ground biomass has been increasing at 0.62 Mg C Ha <sup>-1</sup> Y <sup>-1</sup>				
Robinson et al., 2007	Tropical Forests					68% higher amounts of below-ground biomass than previously estimated		
Paoli and Curran 2007	Borneo	Ever-wet	Mature/old growth	Above-ground biomass increment 5.8-23.6 Mg Ha <sup>-1</sup> Y <sup>-1</sup>	Annual fine litter input 5.1-11.0 Mg Ha <sup>-1</sup> Y <sup>-1</sup>			Total amounts of annual NPP related to soil fertility – phosphorus
Whigham et al., 1991	Yucatan, Mexico	Semi-evergreen	Early secondary		A hurricane can increase dead and downed coarse woody debris by 50%			
Wilcke et al., 2004	Ecuador	Montane	Mature/old growth		9.1 Mg biomass Ha <sup>-1</sup>			

### Aboveground biomass

Aboveground biomass is generally derived from field inventory and forest cover data, extrapolated to forest biomass. Uncertainties in the estimates of both biomass and forest cover contribute to a wide range of estimates of carbon stocks in the tropics (Houghton, 2005). Many analysts use the FAO estimates of aboveground biomass. These estimates are derived from national data provided by each country. Since countries often use different inventory systems and methods, comparisons between countries can be difficult. For example, the increase in biomass estimates in tropical forests of Latin America and tropical Africa seen in FAO data from the 1980s to the 2000s is most likely attributed to more forests being inventoried (Houghton, 2005).

Biomass estimates also vary widely because different tropical forests allocate biomass in different ways in response to environmental conditions, and forest composition and structure. Some of variability, however, derives from factors related to how the data is collected, particularly data that are used to extrapolate from ground measurements to forest biomass. For example, measurements taken at the buttresses of individual trees and then extrapolated to total tree biomass have tended to inflate estimates of biomass in some past studies (Malhi et al., 2004). Table 2 highlights some of the historical variability in above ground biomass estimates.

**Table 2** Estimates of total biomass from various studies in tropical forests (mg dry weight per ha)

Above Ground Biomass		Below Ground Biomass	Reference
Living	Total		
413.4	425.2	104.0	Russell, 1983
406.3		67.0	Klinge and Rodrigues, 1973
358.0	396.2		Delaney et al., 1997
347.7	371.2	56.5	Grimm and Fassbender, 1981
346.0	395.0		Delaney et al., 1997
343.0	351.0		Overman et al., 1994
314.0	353.8		Delaney et al., 1997
306.2	348.0		Uhl et al., 1988
296.0	308.0		Delaney et al., 1997
285.0	325.0		Brown et al. 1995
267.0	320.0	68.0	Salomao et al. 1996
264.0		35.4	Nepsted, 1989
221.0	247.3	58.2	Saldarriaga et al., 1988
242.2	264.6	46.0	Fearnside et al., 1993
140.0	155.2		Delaney et al., 1997

Source: Modified from Houghton et al., 2001

In response to elevated CO<sub>2</sub>, many models predict increased productivity (Laurance et al., 2004; Lewis et al., 2004), both in semi-evergreen forests of the Amazon and central Africa. Feeley et al. (2007) found, however, that on ever-wet sites in Panama and Malaysia, stem growth rate actually decreased from 1981-2005 largely due to increased nighttime temperature, decreased total precipitation, and increased cloudiness.

Decreases in stem growth rate may not be indicative of overall productivity decline, however. Trees could be shifting their allocation of resources from stem growth to root growth, leaf production and/or reproduction (LaDeau and Clark, 2001). Nevertheless, even if overall productivity is increasing, decreased stem growth could affect carbon sequestration if, for example, the residence time of carbon in fine roots, leaves, flowers, or fruits is shorter than in coarse woody material (Pregitzer et al., 1995).

Studies have found large differences in productivity between Southeast Asian tropical forests and those in the neotropics. In a meta analysis of 39 diverse neotropical forests (dry to wet, lowland to montane, nutrient-rich to nutrient-poor soils), total net primary productivity (NPP – above and below-ground) ranged from 1.7 to 11.8 Mg C/ha/yr (lower bounds) and from 3.1 to 21.7 Mg C/ha/yr (upper bounds) (Clark et al., 2001). In a tropical Asian ever-wet forest in southwest Borneo, however, Paoli and Curran (2007) found that above ground NPP alone ranged from 11.1 to 32.3 Mg C/ha/year, which implies that total NPP is much higher than in neotropical forests. Paoli and Curran (2007) also found that the spatial pattern of productivity in the lowland Bornean forests was significantly related to soil nutrients, particularly phosphorus. It is important to note that almost all the work cited here is from semi-evergreen and ever-wet forests of the Amazon, Central America, and Malaysia/Borneo. Little work has been done in other regions on this topic, especially in dry deciduous and montane forest types.

### **Belowground biomass**

Measuring belowground biomass is very difficult because roots are embedded in the soil. Not only is uncertainty in inventory data problematic for belowground biomass estimates, but direct measurement is often very difficult even with the most thorough root collection (Robinson, 2007). Attempts to remove entire trees and their root systems tend to underestimate root biomass because many of the fine roots remain in the soil. Current estimates of root masses could be understated by as much as 60% according to Robinson (2007), who provides adjusted values for biomes to reflect this discrepancy. These findings suggest that root mass for tropical forests worldwide could contain up to 49 more Pg of carbon than found in previous studies, with a subsequent increase in total carbon sink of 9% for tropical forests (Robinson, 2007). More belowground biomass could account for some of the “missing” global carbon sink and has implications for soil carbon estimates as well.

Understanding how belowground carbon allocation varies with soil and topographic conditions and across different climates is crucial to linking the different carbon pools in forests. Belowground biomass allocation can differ significantly both

spatially and temporally in tropical forests. Spatial variation in belowground fine root biomass for an ever-wet forest at La Selva research station in Costa Rica was similar to that of studies done in other tropical and temperate forests (Espeleta and Clark, 2007). Higher fine root biomass in the soil profile was associated with less fertile oxisols, lower in phosphorous, and with less soil water availability across a landscape gradient, while lower fine root biomass was associated with greater fertility and soil water availability in the soil profile. Espeleta and Clark (2007) produced the first belowground dataset for tropical forests to sufficiently assess temporal variation of fine root stocks. They found that sites on slope crests had greater live and dead fine-root variation in turnover due to changes in soil water content and its effect on nutrient acquisition. Drier years led to increased litter fall, and tree and root mortality. This has implications for how belowground biomass allocation and nutrient cycling may be impacted in a changing climate. If tropical forest soils dry as predicted by many models (e.g., Cox et al., 2000; Friedlingstein et al., 2006; Notaro et al., 2007), then fine roots located in the driest portions of the soil profile should die. If water stress does not lead to mortality, then plants should respond by allocating more root biomass to wetter areas of the soil profile.

### **Epiphytes, litter and logs**

There have been numerous studies on the role of coarse woody debris in temperate forests – particularly old growth (Harmon et al., 1986). However few such studies have been done for tropical forests. Dry deciduous and semi-evergreen forests might have larger proportional loads of coarse woody debris than ever-wet and montane forests because of proneness to hurricanes and fire and greater impacts from swidden/fallow cultivation systems. For example, Eaton and Lawrence (2006) found that in the northern Yucatan, the largest amounts of downed debris were recorded post land clearance (88% of above ground biomass). Studies by others have shown that hurricanes can create large amounts of coarse debris, not directly, because most vegetation survives and re-sprouts, but indirectly through susceptibility to fire (Whigham et al., 1991).

Studies of coarse woody debris in ever-wet forests are also rare. One study in Costa Rica found no difference in standing dead and downed wood (> 10 cm in diameter) in relation to topography and soil, but that overall it contributed to 33% of the above-ground biomass, with a turnover of about 9 years (Clark et al., 2002). In a semi-evergreen forest in the Brazilian Amazon, downed coarse woody debris was recorded between 50-55 Mg biomass per ha (Keller et al., 2004). For ever-wet forests in Costa Rica (Clark et al., 2002) and the Peruvian upper Amazon (Baker et al., 2007) stocks were about the same (22 and 24 Mg C per ha respectively or 46 and 50 Mg biomass per ha). In an Ecuadorian montane forest, Wilcke et al. (2004) found much lower woody debris biomass stocks (9 Mg biomass per ha) but it was highly variable and represented only 4% of the total estimated carbon in the forest.

Litter production in tropical forests is likely to increase in an elevated CO<sub>2</sub> environment as it is linked to higher respiration rates (Sayer et al., 2007). Litter production in the tropics, and indeed aboveground productivity, is related to soil

nutrients, especially phosphorous, in addition to carbon (Paoli and Curran, 2007). CO<sub>2</sub> enrichment tends to have a positive effect on plant growth up to a certain point before plants begin to exhaust other resources and reach a limit of enhanced growth, at which point litter production levels off.

### Soil carbon

Most soil carbon in tropical forests is located in the uppermost layers where root density is generally the highest. In a soil respiration measurement experiment comparing sites in Paragominas, Brazil (semi-evergreen) and La Selva, Costa Rica (ever-wet), Schwendenmann and Veldcamp (2005) found that more than 75% of the CO<sub>2</sub> was produced in the upper 0.5 m (including the litter layer) while less than 7% came from soil below 1 m depth. CO<sub>2</sub> production was positively correlated with both temperature and soil moisture in the top 0.5 m (Schwendenmann and Veldcamp, 2005). In the Paragominas site, beyond 2 m in soil depth CO<sub>2</sub> production increased greatly with increasing temperature (Schwendenmann and Veldcamp, 2005). Nevertheless, this is still a much lower amount of flux than in the upper layers. The increases in CO<sub>2</sub> production observed by Schwendenmann and Veldcamp (2005) indicate a strong positive feedback between ecosystem warming and CO<sub>2</sub> flux from moist tropical forest soils, but further studies need to verify this.

This study also highlights how differences in local climate, soil, and forest type can affect soil carbon flux. Paragominas is a tropical deciduous forest with a long dry season. Its forests have deep roots to a depth of at least 18 m (Nepstad et al., 1994) that enable them to extract water stored at greater depths. Active soil water extraction occurs with root respiration, which can explain the high CO<sub>2</sub> production observed in the deep soil at the site in Paragominas (Schwendenmann and Veldcamp, 2005). In contrast, the forest at La Selva does not experience an intense seasonal drought and the water content below 0.75 m is always above field capacity. It also has a low root biomass below 2 m (Veldkamp et al., 2003). The contribution of root respiration to CO<sub>2</sub> produced beyond 2 m in depth at La Selva is minimal. Deep soil CO<sub>2</sub> at La Selva is principally from decomposition of soil organic carbon and/or dissolved organic carbon by soil microbes (Schwendenmann and Veldcamp, 2005). The sheer contrast in CO<sub>2</sub> production at different depths of different soil and forest types highlights the complexity of soils and the need to further examine microbial and plant biochemical processes in deeper soil layers over longer periods (see Chapter 2 for further details).

The dynamic changes in the composition of the soil microbial community in response to inputs of organic matter may increase soil respiration rates and thus drive soil carbon losses in the form of carbon dioxide to the atmosphere (Cleveland and Townsend, 2006; Cleveland et al., 2007). Since many climate models predict further soil drying and increased litter fall in tropical forests (e.g., Cox et al., 2000; Friedlingstein et al., 2006; Notaro et al., 2007), understanding the role of the soil microbial community and its function within the litter layer, belowground biomass and soil carbon is key. Changes in climate, the concentration of CO<sub>2</sub> in the atmosphere, and the nutrient content of litter could all have an effect on soil biota and decomposition rates (Coley, 1998).

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The sheer contrast in CO<sub>2</sub> production at different depths of different soil and forest types highlights the complexity of soils and the need to further examine microbial and plant biochemical processes in deeper soil layers over longer periods.

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## BIOTIC DRIVERS OF UPTAKE AND RELEASE

Since the early 1980s, only three studies have analyzed land surface-atmosphere interactions in tropical forest ecosystems: the Anglo-Brazilian Climate Observation Study (ABRACOS; 1990–95); the Large-scale Biosphere/Atmosphere Experiment in Amazonia (LBA; 1996–2003); and the GEWEX Asian Monsoon Experiment (GAME; since 1996) (Nightingale et al., 2004). All three studies were conducted in semi-evergreen forests. It is difficult to model carbon flux and productivity in tropical forests due to their structural and age complexity and species composition. As a result, few ecosystem process models have been developed, parameterized, and applied within tropical forest systems (Nightingale et al., 2004). Nevertheless, it is a reasonable assumption that rising temperatures will increase the rate of most if not all biochemical process in tropical plants and soils (Lloyd et al., 1996). Therefore, it is essential to understand how carbon is taken up by plants and the pathways of carbon release, and how increasing temperatures could affect these processes and the balance between them.

### Photosynthesis and autotrophic respiration

Photosynthesis is the process through which plants take up carbon in the form of carbon dioxide ( $\text{CO}_2$ ). Specifically, it requires  $\text{CO}_2$ , sunlight, and water as inputs to produce glucose (carbohydrates), oxygen, and water. If the carbon uptake of photosynthesis exceeds the carbon efflux of respiration, intact forests are thought to remain a carbon sink (Phillips et al., 2008). However, the increases in productivity observed in Amazonian and Central African semi-evergreen and tropical forests over the past few decades by Phillips et al. (2008) and Lewis et al. (2009) cannot continue indefinitely. Lewis et al. (2009) estimate that one fifth of the  $\text{CO}_2$  currently produced globally by land conversion and industrial emissions is absorbed by the tropical forest regions through increased productivity. However, if  $\text{CO}_2$  is the cause for this increased productivity, then trees will eventually reach a saturation point and become limited by some other resource (Phillips et al., 2008). Thus, it is critical to consider the role of other biogeochemical cycles in relation to carbon.

Many future climate scenarios predict soil drying in Amazonia and a general reduced capacity of the ecosystem to take up carbon (Friedlingstein et al., 2006; Notaro et al., 2007). Understanding how tropical forests respond to water stress could be important because canopy-to-air vapor deficits and stomatal feedback effects could determine how tropical forest photosynthesis responds to future climate change (Lloyd et al., 1996).

### Heterotrophic respiration and decomposition

Respiration requires oxygen, carbohydrates, and water to release energy,  $\text{CO}_2$  and water. Autotrophic respiration occurs when plants release  $\text{CO}_2$  during biochemical processes, such as growth and production of chemical defenses. Heterotrophs (e.g., animals) also contribute to  $\text{CO}_2$  release in a similar process. Like photosynthesis, respiration is linked to temperature fluctuations and other environmental factors (Phillips et al., 2008).

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Decomposition is a type of respiration in which dead organic matter, oxygen, and water are converted into energy, CO<sub>2</sub>, and water. Barring poor access to moisture and oxygen, decomposition in the humid tropics tends to be rapid, which limits the accumulation of detritus on the forest floor (Ricklefs, 2001; Vandermeer and Perfecto, 2005). Where moisture stress or oxygen stress inhibit aerobic respiration, however, detritus can accumulate, such as in peat swamps and other poorly drained areas or certain areas of tropical dry forests. When oxygen stress limits aerobic respiration, microbes and fungi responsible for decomposition rely on anaerobic respiration – a less efficient method of respiration in which methane is often a byproduct.

As tropical forest soils become drier, litter decomposition and its release of CO<sub>2</sub> from soil may slow in response to less water availability. However, Cattânio et al. (2002) found that greenhouse gas release in the form of methane, which has a higher global warming potential than CO<sub>2</sub>, increased as soils dried in plots where water was excluded. This is surprising, as methane production requires anaerobic microsites that are uncommon in dry soils. Dry plots in their study had more litter and woody debris; there was also anecdotal evidence of increased termite activity, which may explain the release of methane (Cattânio et al., 2002). Indeed, changes in soil moisture stand to not only affect the response of plant species and communities, but also the population dynamics of animals, fungi and microbes, which in turn will have impacts on herbivory and decomposition. Thus, it is important to remember that changes in ecosystems rely on the interaction of all of its components, not just a few.

#### **DISTURBANCE: ABIOTIC DRIVERS OF UPTAKE AND RELEASE**

Disturbance is a natural part of any ecosystem, to which most organisms have some form of adaptation. Tropical forests experience tree mortality from old age, earthquakes or storms, which open up the forest floor to light and allow younger trees to attain the canopy. Downed trees occasionally survive and continue growing, however, and fill in gaps themselves (Vandermeer and Perfecto, 2005). When trees die, they decompose and release CO<sub>2</sub> and nutrients to the soil and atmosphere. Nutrients may be taken up quickly by other plants, stored in soil for a period of time, or leached from the system during rain events. In large scale disturbances, especially fires, landslides, land clearance, or logging, large amounts of nutrients are lost from the ecosystem. It may take hundreds of years to recover from this nutrient loss. At the same time, land-use conversion to non-forest uses, such as farms and cities, releases carbon to the atmosphere, further altering the carbon budget of the landscape.

Many studies use old growth sites that have not experienced major disturbances for a long period of time (e.g., Malhi and Phillips, 2004). This has led to unexpected results in carbon flux measurements. In one 3-year study of old growth forests in the Amazon, carbon was released in the wet season and taken up in the dry season, in opposition to the seasonal cycles of both tree growth and model predictions (Saleska et al., 2003). This disconnect was attributed to decomposition and soil moisture availability, transient effects of recent disturbance. This has important implications for carbon budgeting in the Amazon. If studies tend to be concentrated in

undisturbed, old growth forests versus recently or regularly disturbed sites, predictions of future carbon sequestration rates are likely to be overestimated (Saleska et al., 2003).

### **Drought and El Niño-Southern Oscillation (ENSO) events**

ENSO events and droughts are part of the planet's natural climate cycles. Although there has been much research into ENSO events and their effect on droughts in the tropics, droughts can be independent of ENSO events. Combined climate-carbon cycle models predict that the Amazon forests are vulnerable to both short- and long-term droughts (Saleska et al., 2007). When water is initially limited, vegetation responds by reducing transpiration and photosynthesis, which in turn reduces the amount of water recycled to the atmosphere. However, satellite observations of the Amazon showed that there was a large-scale "green up" (re-leafing) of intact semi-evergreen forests in response to a short, intense drought in 2005 (Saleska et al., 2007). This inconsistency with model predictions may be due to the fact that forests were actually not water limited and were able to use deep roots and hydrologic redistribution to access soil water during the drought. In addition, the increased available sunlight due to reduced cloud cover allowed the plants to respond with more growth (Saleska et al., 2007). At least in the short-term, tropical forests may be resilient to drought. However, this may be offset by increased vulnerability to fire after both short- and long-term droughts (Saleska et al., 2007; Nepstad et al., 2007).

Tropical rainforests in the Amazon and southeastern Asia typically suffer from droughts during ENSO events. These droughts are more severe during strong El Niño years (Lyons, 2004). How ever-wet forests and semi-evergreen forests respond to drought varies. In one study of the ever-wet forests of Borneo, where droughts are rare, mortality increased during strong El Niño years due to severe drought, while semi evergreen forests experienced relatively little change (Potts, 2003). In addition to forest type, position in the landscape, soil texture and rooting depth play a role in the vegetation's response to drought (Sotta et al., 2007). For example, temporarily flooded valleys and lowlands often receive drainage from upslope areas and are able to retain moisture longer than uplands (Ashton, 1992; Ashton et al., 1995; Grogan et al., 2003; Ediriweera et al., 2008). Areas with finer soil textures retain more water for longer time periods than those with coarser textures. Texture can vary within the soil profile, which means that the texture of soil at lower depths could be an important indicator for a site's water retention capacity during droughts (Grogan et al., 2003; Sotta et al., 2007). In addition, the location of roots within the soil profile determines where a plant can take up water. During drought events, the surface tends to dry first, putting plants with deeper roots or the ability to quickly respond to drought by allocating root growth to deeper soils at an advantage (Sotta et al., 2007). Increased water stress during drought is linked to higher tree and liana mortality, which suggests that more carbon will be released through decay and increased probability of fire (Nepstad et al., 2007). These differing responses are significant because ENSO events are expected to become more frequent in response to the greenhouse effect (Tsonis et al., 2005).

### Wind and rain

Wind throw and snap-off of trees from winds can occur in a variety of forms from large landscape level effects (hurricanes and typhoons) to more landscape-specific convectional windstorms that affect multiple trees (stand scale) to individual wind throw and branch breakage (Whigham et al., 1999). Most winds come with rain, either before or after. Rain-soaked soils are less firm, and roots insecure, making trees more prone to windthrow.

Seasonality provides another axis for differentiation. Subtropical forests and regions more than 10 degrees north or south of the equator experience greater variation in seasonality, and therefore stronger trade winds, monsoons (and hurricanes), particularly on the eastern sides of continents (e.g. Honduras, Belize, Yucatan-Mexico, Guatemala, Nicaragua, southeastern Africa, Madagascar, Vietnam, eastern coast of the Philippines, southeast China, the Caribbean islands, the southwest Pacific Islands, northeast Australia) (Whigham et al., 1999). These regions can be exposed to periodic large scale wind events which the forests are adapted to – mostly through vigorous re-sprouting (Whitmore, 1989; Whigham et al., 1991; Eaton and Lawrence, 2006). Most forests in these regions would be considered semi-evergreen or dry deciduous – meaning that periods of drying can promote fire for land clearance. In fact, many swidden systems are cleared during the dry season and then burned prior to the rains to take advantage of the pulse of nutrients for crop cultivation in the wet season.

In the more equatorial regions where ever-wet forest dominates, winds often occur with the onset of rains through vigorous frontal or convectional thunderstorms that can knock over large swaths of forest with strong down drafts (Whigham et al., 1999). On steeper and often younger more erosion-prone hills and mountains, large amounts of rain can cause landslides, riparian flooding and bank erosion (e.g. in the Andes, Central American Cordillera, central ranges of Sumatra, Borneo, and Malay Peninsula). All large scale wind and rain events are episodic and occur at relatively long time intervals that are difficult to predict. However, they drive the successional dynamic of forests, and therefore by implication, the above- and below-ground carbon stocks. Little to no work has been done on assessing and including this dynamism in the development of regional carbon models predicting future change. The assumption is that small-scale disturbances in old-growth forests will remain the dominant phase of growth.

### Fire

Fires in tropical forests are typically the result of drought and human land management practices (Bush et al., 2008). Indeed, fire is thought to be a more imminent threat to tropical forests than climate change (Barlow and Peres, 2004; Nepstad et al., 2004; Bush et al., 2008). In contrast to other biomes, such as certain jack pine (*Pinus banksiana*) boreal forests, where fire events tend to be naturally occurring, humans have always been the primary ignition source of fires in tropical forests. In fact, natural fire in the Amazon has been so rare since the mid-Holocene that the presence of charcoal in soil is taken as an indicator of human activity (Bush

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et al., 2008). Under normal moisture conditions, the likelihood of fire decreases exponentially with distance from roads and clearings (Cochrane and Laurance, 2002). This supports the view that fire is a direct result of human activity in tropical systems. If drought becomes more common in tropical forests as some climate models predict (e.g., Cox et al., 2000), the likelihood of human-induced fires escaping and impacting large portions of the landscape increases. This was seen during the ENSO-induced drought in tropical Indonesia and Amazonia in 1997-1998 where drought caused many human-ignited fires to escape and become wildfires (Bush et al., 2008).

In addition to climate, the impact of fire on tropical forests is also highly linked to forest structure. Nepstad et al. (2007) found that mortality of large trees and lianas following an experimental drought increased. Large trees not only store significant amounts of carbon, but also provide shade, which helps to keep litter moist. The absence of this shade dries out the litter layer and the dead lianas become ladder fuels, thus increasing the probability that an escaped fire will burn the litter layer and reach the canopy (Nepstad et al., 2007). This in turn is likely to impact what types of plants can regenerate and colonize after a fire.

### Herbivory

The effects of elevated atmospheric CO<sub>2</sub> and global climate change on herbivory and other plant/animal interactions in tropical forests are not well understood. Little research has been done in this area. One seminal piece, Coley's "Possible effects of climate change on plant/herbivore interactions in moist tropical forests" (1998) addressed the interdependent roles of climate change and herbivory in tropical forests. More research into how climate change and elevated greenhouse gases will affect herbivore-plant and other predator-prey dynamics is needed. Indeed, although the Coley study is a core research paper on this topic, even this study is not adequately supported by direct experimentation.

More than 70% of herbivory occurs on young leaves. Herbivores may be susceptible to climate change as plant growth patterns are altered in response to elevated temperatures and CO<sub>2</sub> and increases in the length of the dry season. Although CO<sub>2</sub> increases tend to enhance plant growth rates, drought stress reduces plant growth (Coley, 1998). The net effect on herbivore activity is uncertain.

In elevated CO<sub>2</sub> experiments, the nutritional quality of leaves generally declines due to a 10-30% dilution of nitrogen. This effect is less drastic, however, in plant species that associate with nitrogen fixers. Elevated CO<sub>2</sub> reduces nitrogen-based defenses (e.g., alkaloids) and causes an increase in carbon-based defenses (e.g., tannins). As leaves exhibit lower nutritional value, herbivory may increase substantially to compensate. Furthermore, increased herbivory could affect plant populations, perhaps favoring species with higher chemical and physical defenses or nitrogen fixation capabilities (Coley, 1998). Although one intuitively might expect severe drought to lead to a decrease in herbivore populations, they may indeed increase due to decreased pressure from predators and parasitoids. This could ultimately result in more frequent insect outbreaks (Coley, 1998).

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If drought becomes more common in tropical forests as some climate models predict, the likelihood of human-induced fires escaping and impacting large portions of the landscape increases.

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Future research into plant/herbivore and other plant/animal interactions could include attempts to identify which plant taxa are likely to become more prevalent in future climate models. In addition, more extensive research could be conducted on the effects of changes in plant secondary compounds (e.g., tannins) on herbivore development and aquatic chemistry and food webs in the tropics. There are few controlled experiments on predator/prey interactions under conditions of future climate models, despite the fact that such experiments could shed light on tropical ecosystem resiliency to climate change. Both top-down and bottom-up approaches to predator-prey modeling could substantially increase our knowledge in this realm.

## **CLIMATE CHANGE IMPACTS ON TROPICAL FOREST DYNAMICS**

### **Increased productivity versus increased respiration**

Old-growth tropical forests are experiencing accelerated stand dynamics and increasing biomass (Malhi and Phillips, 2004; Lewis et al., 2009). Studies have shown that there has been a net increase in biomass in recent decades in Amazonian and Central African semi-evergreen and ever-wet forests. Several studies have addressed methodological challenges in measuring biomass (Baker et al., 2004; Chave et al., 2004). According to new estimations by Malhi and Phillips (2004), the net carbon sink of intact old growth forests of Amazonia is  $0.9 \pm 0.2$  Mg C per ha per year. Applying this rate to the area of moist forest in Amazonia, the Amazon rain forest is thought to sequester nearly 0.6 Pg C per year.

Like many ecological processes, biomass growth does not occur in isolation. Forest turnover rates in Amazonia have accelerated (Phillips et al., 2004; Laurance et al., 2004). In particular, the greatest increases in turnover rates have occurred on more fertile soils in western Amazonia. This increase in recruitment has been greater than the increase in mortality, which has actually lagged behind this acceleration in growth (Phillips et al., 2004). Similarly, Laurance et al. (2004) found that forests of central Amazonia have experienced changes in dynamics and composition that are not due to any detectable disturbance. In a network of 18 permanent study plots, not only have mortality, recruitment, and growth rates increased over time, but 27 of 115 relatively abundant tree genera have changed significantly in population density or basal area. Furthermore, genera of faster-growing canopy and emergent tree species – not necessarily pioneer species – are increasing in dominance or density, while genera of slower-growing subcanopy tree species are declining. Elevated CO<sub>2</sub> is one of several possible explanations for changes. What is certain, however, is that these changes in dynamics and composition could have important impacts on the carbon storage and biota of Amazonian forests (Laurance et al., 2004).

Most climate models and forest carbon balance models do not take forest composition into account (Phillips et al., 2008). Forests with accelerated or “faster” dynamic have less biomass due in part to ecophysiological differences in plant growth. For example, fast growing species have less dense wood, and therefore less stored carbon, compared to slow-growing species which have denser wood, with

more carbon. Early successional forest therefore has less carbon stored than late successional forest (Phillips et al., 2008). A summary of how increasing CO<sub>2</sub> concentrations may or may not affect tropical forest growth is provided in Table 3 (Malhi and Phillips, 2004).

**Table 3 Arguments to expect, or not, substantial effects of increasing CO<sub>2</sub> concentrations on tropical forest growth and carbon balance**

zero or negative effect on growth and carbon storage	positive effect on growth and carbon storage
<b>leaf level</b>	
increased respiration and photorespiration caused by rising temperatures warming temperatures lead to increased evaporative demand, inducing stomatal closure and reducing photosynthesis increased emissions of volatile hydrocarbons at higher temperatures consume assimilated carbon	direct fertilization of photosynthesis by high CO <sub>2</sub> reduced photorespiration caused by high CO <sub>2</sub> improved water-use efficiency caused by high CO <sub>2</sub> photosynthetic rates increase with moderate warming optimum temperature for photosynthesis rises with rising CO <sub>2</sub>
<b>plant level</b>	
plants are often saturated with respect to non-structural carbohydrates? plant growth limited by nutrients other than carbon (N, P, K, Ca) rising soil temperatures increase soil acidification and mobilize aluminium, reducing soil nutrient supply plant carbon balance is limited by respiration costs rather than by photosynthesis gains? acclimation (downregulation of photosynthesis) limits any response to increasing CO <sub>2</sub>	excess carbon may be used preferentially above ground to acquire rate-limiting resource (light) by investing in wood excess carbon may be used preferentially below ground to acquire rate-limiting resource (nutrients) through fine root development or supporting P-scavenging mycosymbionts rising temperatures increase soil mineralization rates and improve nutrient supply
<b>stand level</b>	
biomass ultimately limited by disturbance (e.g. windthrow risk) rather than by resources forest canopies are close to physical limits of forest structure which cannot be increased (e.g. maximum tree height is limited by hydraulics or mechanics) faster growth and turnover may favour disturbance-adapted taxa, with less dense wood lianas may benefit from increased CO <sub>2</sub> and disturbance, limiting biomass gains by trees mortality rates increase because of climatic warming and/or drying, or increased climatic variability climatic drying combined with forest fragmentation and degradation lead to increased fire frequency	faster growth leads to some biomass gains, with mortality gains lagging rising CO <sub>2</sub> improves water-use efficiency and reduces tension in the water column, allowing an increase in maximum tree height for a given cross-sectional area faster growth and turnover may prevent stand dominance by senescent 'over-mature' trees with high respiration costs, creating positive feedback on stand-level growth rates improved forest water balance leads to reduced drought mortality and fire incidence

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It is possible that the most significant impact of rising CO<sub>2</sub> levels in the atmosphere thus far has been an increase in fire frequency, thus altering the boreal forest age-class distribution.

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Source: From Malhi, Y., Phillips, O.L., 2004. Tropical forests and global atmospheric change: A synthesis. Philosophical Transactions of the Royal Society of London Series B-Biological Sciences 359, 549-555. Reprinted with permission.

Tropical forests are resilient to many types of environmental change. However, given the human footprint in many of these forests, the expected resiliency may not materialize (Cowling and Shin, 2006). Evergreen rain forests have dominated the Amazon Basin since the last glacial maximum (Beerling, 2006). Historically, climate change has driven biome shifts in transition or ecotonal zones, while CO<sub>2</sub> changes

have led to increased carbon storage (Beerling, 2006). Many transition zones (e.g., montane forests) and tropical seasonal forests are areas that have experienced rampant deforestation and other types of land-use change in the past century. This may yield some insight into how tropical forests might change in both composition and range in response to climate change (Malhi and Phillips, 2004). For example, a warmer climate could drive low elevation forests to higher elevations or extend the range of tropical seasonal forests. However, if there is more deforestation in these seasonal and dry areas, there may be fewer species available that can migrate and adapt to warmer climates with drier soils (Malhi and Phillips, 2004).

Many models have predicted decreased forest cover and soil drying over Amazonia in response to the radiative effect of rising CO<sub>2</sub> concentrations in the atmosphere (Friedlingstein et al., 2006; Notaro et al., 2007). What happens to soil carbon pools and the dead biomass from this reduced forest cover is of great importance to researchers studying carbon fluxes under climate change. The fate of these carbon pools under the most extreme scenario modeled – wide-spread tree die-off – depends on drought conditions and elevated soil respiration under higher temperatures (Cox et al., 2000). As air temperature rises, respiration increases, while carbon uptake from photosynthesis continues until it reaches some threshold. In short, the current carbon sink that intact tropical forests provide cannot continue in the same manner indefinitely. How this carbon balance could change, apart from the immediate threats of land use change, habitat fragmentation and fire, is uncertain.

Phillips et al. (2008) provide three scenarios about the future of this carbon sink in the Amazon base on an extensive network of research sites: mature Amazonian forests will either (i) continue to be a carbon sink for decades (Cramer et al. 2001); (ii) quickly become neutral (i.e., uptake equals release) or a small carbon source (Cramer et al., 2001; Körner, 2004; Laurance et al., 2004) or (iii) become a mega-carbon source (Cox et al., 2000; Lewis, 2006). The difference between the annual stand level growth (uptake: 2%) and mortality (release: 1.6%) of Amazonia is currently estimated to be 0.4%, which is just about enough carbon sequestered to compensate for the carbon emissions of deforestation in the region (Phillips et al., 2008). This means that either a small decrease in growth or a small increase in mortality in mature forests could convert Amazonia from a sink to a source of carbon (Phillips et al., 2008). Better estimates are needed of the amount of mature, secondary and disturbed forests in the Amazon in order to better predict changes in carbon storage trends and the threat of release of this terrestrial sink.

### Changes in precipitation amounts and patterns

Some models have predicted that reduced forest cover and soil drying over Amazonia, in response to the radiative effect of rising CO<sub>2</sub> concentrations in the atmosphere, will result in a reduction in the land's capacity to take up carbon (Friedlingstein et al., 2006; Notaro et al., 2007). In simulation modeling of ecosystem threshold responses to changes in temperature, precipitation, and CO<sub>2</sub>, Cowling and Shin (2006) found the 'natural,' intact Amazonian rainforest to be resilient to environmental change, particularly to decreases in temperature and precipitation.

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The difference between the annual stand level growth (uptake: 2%) and mortality (release: 1.6%) of Amazonia is currently estimated to be 0.4%, which is just about enough carbon sequestered to compensate for the carbon emissions of deforestation in the region.

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However, they also warn that humans have changed these forests so quickly in the past several decades that the resiliency of the Amazonian rain forest is at risk (Cowling and Shin, 2006). Asian ever-wet forests are thought to be considerably more sensitive to drying conditions (Paoli and Curran, 2007). However, the interactions with other human disturbance factors such as land clearance, edge effects and fragmentation, and fire need to be considered and could have important negative feedback influences (Leighton and Wirawan, 1986).

### **Land use change**

Human intervention through deforestation and forest degradation has been the leading cause of perturbation to the carbon cycle in tropical forests (Houghton, 1991a; Sampson et al., 1993). As a result, by the year 2050, the tropics could be a source of atmospheric CO<sub>2</sub> (Sampson et al., 1993). Land use change is perhaps the most imminent threat to the ecosystem services that tropical forests provide. It is believed that land use change could lead to the release of 40–80 Pg C per year over the next 50 years (Nightingale et al., 2004).

### *Deforestation*

Deforestation affects the carbon balance of tropical forests and climate feedback cycles in two principal ways: carbon emissions from deforestation and the albedo effect of deforested lands (Bala et al., 2007; Ramankutty et al., 2007). Moreover, for every ton of carbon released to the atmosphere through deforestation, 0.6 additional tons of carbon are released through degradation of the remaining forest (Houghton, 1991a). However, current estimates of carbon emissions from tropical deforestation vary greatly and are difficult to compare due to differences in data sources, assumptions, and methods (Ramankutty et al., 2007). Developing and incorporating multiple variables into new and existing ecosystem models for tropical forests is essential to determining carbon fluxes and future effects of deforestation and climate change (Nightingale et al., 2004). In order to fully quantify the carbon emissions from tropical deforestation, one must account for initial carbon stock of vegetation and soils, influence of historical land use, rates and dynamics of land-cover changes, methods of land clearing and the fate of the carbon from cleared vegetation, response of soils following land-cover change, and the representation of processes in ecosystem and climate models used to integrate all of these components (Ramankutty et al., 2007).

While it is a fact that deforestation releases CO<sub>2</sub> to the atmosphere, which in turn has a warming effect on climate, there is another important piece of the deforestation equation that some models neglect. Deforestation comes with biophysical effects on climate, such as changes in land surface albedo, evapotranspiration, and cloud cover. Simulations out to 2150 by Bala et al., (2007), using a three-dimensional model representing physical and biogeochemical interactions between land, atmosphere, and ocean, found that at a global level, deforestation has a net cooling effect on climate. This is because the net cooling influence of changes in albedo and



evapotranspiration outweigh the warming effects associated with carbon release (Bala et al., 2007). It is noteworthy that the model predicted different effects associated with the deforestation of tropical vs. temperate and boreal forests. According to the model results, afforestation in the tropics would be beneficial because of the greater role of tropical forests in increasing evapotranspiration, CO<sub>2</sub> sequestration, and cloud cover and thus reducing the heating impacts of global warming. In contrast, deforestation of higher latitude boreal forests would greatly increase albedo relative to evapotranspiration and CO<sub>2</sub> sequestration having an overall positive effect on climate (Bala et al. 2007). It must be emphasized that this is a single study involving simulations so caution needs to be used in interpreting these results.

### *Agriculture*

Expanding crop and pasture lands have a profound effect on the global carbon cycle as tropical forests typically store 20-100 times more carbon per unit area than the agriculture that replaces them (Houghton, 1991a). In the Amazon, the growing profitability of large-scale industrial agriculture and cattle ranching has led to significant deforestation. This will only increase forest fragmentation and degradation and subsequent climate effects as it continues to expand (Nepstad et al., 2008). The use of fire to clear forested lands may exacerbate changes to carbon cycling since fire fills the atmosphere with aerosols, thereby reducing transpiration (IPCC, 2007).

Within Southeast Asia, the conversion of tropical forests to oil palm plantations is accelerating. This land use change results in a significant net loss of carbon to the atmosphere since the aboveground biomass of oil palm plantations stores less carbon (<36–48 tons C/ha) than tropical primary forests (235 tons C/ha) (Reijnders and Huijbregts, 2008). Including carbon releases for fire, which is the primary method for land clearing, the net carbon loss from the system may be as much as 187-199 tons C/ha. If such fires are of high intensity, there is even greater loss of soil carbon (Reijnders and Huijbregts, 2008). A full life cycle analysis of forest conversion and carbon loss, and then cultivation and production of biofuels from oil palm, puts into question the assertion that oil palm reduces CO<sub>2</sub> emissions (Reijnders and Huijbregts, 2008).

## **CONCLUSIONS AND SUMMARY RECOMMENDATIONS**

Tropical forests account for almost half the gross primary productivity of the world's terrestrial ecosystems. Tropical ever-wet and semi-evergreen forests in the Amazon and southeastern Asia typically suffer from droughts during ENSO events. In the short term, tropical forests may be resilient to drought, but increased susceptibility to anthropogenic fire may negate this. In tropical ever-wet forests, where droughts are rare, mortality may dramatically increase. Seasonal semi-evergreen forests may show little change.

In tropical forest regions humans have caused most fires. If climate model predictions are correct, increased drought will promote the escape of human-caused

fires that will impact large portions of the remaining forest. More work should be done to investigate the negative and positive feedbacks of drought, windstorms, insects/pathogens, fire and humans, and their interactions, on the forest dynamic – and in particular, their effects on carbon.

According to long-term permanent plot data, old growth ever-wet and semi-evergreen forests show increasing biomass in recent decades in Amazonian and Central African forests. This is hypothesized to be in response to increased atmospheric CO<sub>2</sub>.

Uncertainties in estimates of both biomass and deforestation contribute to a wide range of estimates of carbon emissions in the tropics. Better estimates are needed of the amount of mature, secondary, and disturbed forest in order to better predict changes in carbon storage trends. Dry deciduous and montane forests are almost a complete unknown because so little work has been done on these forest types; therefore, much more research needs to be carried out in these areas. Even though in the Americas and Asia many dry deciduous forests have been cleared, significant amounts remain in Africa.

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## Chapter 5

# Carbon Dynamics of Temperate Forests

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### EXECUTIVE SUMMARY

Twenty-five percent of the world's forests are in the temperate biome. They include a wide range of forest types, and the exact boundaries with boreal forests to the north and tropical forests to the south are not always clear. There is a great variety of species, soil types, and environmental conditions which lead to a diversity of factors affecting carbon storage and flux. Deforestation is not a major concern at the moment. The biome is currently estimated to be a carbon sink of about 0.2 to 0.4 Pg C/year, about 37% of the total net terrestrial carbon uptake, disproportionately higher than its representative area, with most of the sink occurring in North America and Europe.

Temperate forests have been severely impacted by human use – throughout history, all but about 1% have been logged-over, converted to agriculture, intensively managed, grazed, or fragmented by sprawling development. Nevertheless, they have proven to be resilient – mostly second growth forests now cover about 40-50% of the original extent of the biome. Although remaining intact temperate forests continue to be fragmented by development, particularly in North America, there is no large-scale deforestation at present, nor is there likely to be in the future. The status of the temperate biome as a carbon reservoir and atmospheric CO<sub>2</sub> sink rests mainly on strong productivity and resilience in the face of disturbance. The small “sink” status of temperate forests could easily change to a “source” status if the balance between photosynthesis and respiration shifts even slightly.

### What do we know about carbon storage and flux in temperate forests?

- Older forests have more carbon stock than younger stands, and mixed species stands in the moist broadleaf and coniferous forest type tend to have higher carbon density than single species stands. Younger stands tend to

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have higher rates of carbon sequestration, as indicated by net ecosystem productivity (NEP), than mid- or older-aged stands, although the data are highly variable.

- The belowground carbon pool of living biomass (primarily roots), roughly estimated to be 5% to 10% of total carbon, is much smaller than the above ground pool; however, this is a tenuous conclusion because the below ground biomass carbon pool is the least studied part of the forest carbon budget.
- Soils contain at least half the carbon in temperate forests and possibly as much as two-thirds; this carbon pool appears to be stable under most disturbances, such as logging, wind storms, and invasive species, but not with land use change. Huge losses can occur when converting forests to agriculture or development.
- Atmospheric pollution, primarily in the form of nitrogen oxides (NO<sub>x</sub>) emitted from burning fossil fuels and ozone (O<sub>3</sub>), is a chronic stressor in temperate forest regions. Because most temperate forests are considered nitrogen-limited, nitrogen deposition may also act as a growth stimulant (fertilizer effect). Under current ambient levels, nitrogen deposition is most likely enhancing carbon sequestration; however, the evidence regarding long-term chronic nitrogen deposition effects on carbon sequestration is mixed.

#### **What don't we know about carbon storage and flux in temperate forests?**

- Data on mineral soil carbon stocks in temperate forests can only be considered approximations at this time as there is very little research on deep soil carbon (more than 100 cm).
- Global circulation models predict that higher concentrations of atmospheric CO<sub>2</sub> will increase the severity and frequency of drought in regions where temperate forests are found. However, there is a great deal of uncertainty about how drought will affect carbon cycles.
- Little is known about how the interactions between temperature, moisture, available nutrients, pollutants, and light influence key environmental variables, such as drought, to affect ecosystem carbon flows.

#### **What are the major influences on carbon storage and flux in temperate forests?**

- There is tremendous variability in carbon stocks between forest types and age classes; carbon stocks could easily be lost if disturbance or land use change shifts temperate forests to younger age classes or if climate change or land use change shifts the spatial extent of forest types. On the other hand, if temperate forests are managed for longer rotations, or more area in old growth reserves, then the carbon stock will increase.

- Temperate forests are strongly seasonal, with a well-defined growing season that depends primarily on light (day length) and temperature. This is probably the most important determinant, along with late-season moisture, of temperate forest productivity and hence carbon sequestration.
- On balance, the evidence regarding nitrogen deposition effects on carbon sequestration is mixed. Under current ambient levels, nitrogen deposition is most likely enhancing carbon sequestration. However, under chronic nitrogen deposition, temperate forests may no longer be nitrogen limited, thus the nitrogen “fertilization” effect will be diminished as other resources become constrained.

### **How might the carbon status of temperate forests change with changing climate?**

- There is evidence of increasing productivity in temperate forests as climate has warmed in the last ~50 years: however, this is confounded by successional dynamics and environmental variables. The atmospheric system has not only experienced changes in temperature, precipitation, and radiation, but in CO<sub>2</sub> concentration and pollutants.
- The few studies that have modeled multi-factor influences on temperate forest net ecosystem productivity or carbon flux have found that combined effects are expected to diminish the effect of CO<sub>2</sub> enrichment alone.
- Natural disturbances, particularly windstorms, ice storms, insect outbreaks, and fire are significant determinants of temperate forest successional patterns. The frequency of stand-leveling windstorms (hurricanes, tornadoes) is expected to increase under a warmer climate in temperate moist broadleaf and coniferous forest regions, so that fewer stands would reach old-growth stages of development.
- If changing climate alters the frequency and intensity of fires, re-vegetation and patterns of carbon storage will likely be affected, particularly in interior coniferous forests of mountains, and the woodlands and pinelands of Mediterranean climates.

**Keywords:** carbon, carbon sequestration, coniferous, deciduous, deposition, disturbance, drought, Europe, fire, hardwoods, hurricanes, insects, nitrogen, North America, pinelands, spodosols, temperate forests, wind, woodlands.

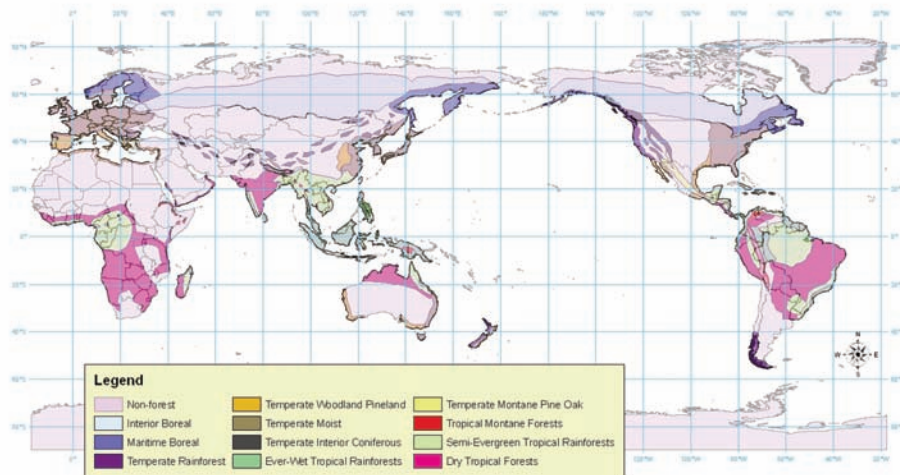
## **INTRODUCTION**

This chapter is a review of the literature on carbon dynamics in temperate forests of Eurasia and North America. It first describes the region, the forest types, and their climatic variations. It then describes the stocks of carbon within the different components of the forest – above ground biomass, below ground biomass, the litter layer, and the soil. The next part is focused on changes among carbon stocks – in

particular understanding the biotic interactions of uptake (photosynthesis) and loss (respiration, decomposition), and then how abiotic influences of disturbance (fire, insect outbreaks, wind, nitrogen deposition, forest management) can affect carbon stocks. We then discuss how changes in climate might impact carbon storage and flux in temperate forests (changes in net primary productivity (NPP) and disturbance regimes). The chapter highlights areas of carbon forest science that we know versus those aspects that we do not know and those in which more work needs to be done.

Twenty-five percent of the world's forests are in the temperate biome, primarily in North America, Europe, Australia, and China, with the remainder scattered throughout the rest of northeast Asia (Japan, Korea), New Zealand and South America (Figure 1; Dixon et al., 1994). They include a wide range of forest types and the exact boundaries with boreal forests to the north and tropical forests to the south are not always clear. There is a great variety of species, soil types, and environmental conditions which lead to a diversity of factors affecting carbon storage and flux.

**Figure 1** Original extent of boreal, temperate, and tropical forest types of the world prior to land clearing



These forests, covering about 10.4 million km<sup>2</sup> (IPCC, 2000; Heath et al., 1993; Dixon et al., 1994), exist in large blocks of forest cover on the eastern and western sides of five continents, broken by extensive areas of prairie, steppe, and desert within the continental interiors. Deforestation is not a major concern at the moment, as global temperate forest area is fairly stable. Historically, these forests have been exploited for timber and charcoal, cleared for agriculture and development, and otherwise heavily impacted by humans (Heath et al., 1993; Nabuurs et al., 2003). Current area is estimated to be around 40-50% of the original extent (Smith et al., 2009; Bryant et al., 1997), with most of the historical loss occurring in Europe, where many countries had lost more than 90% of their forest cover by the late medieval

period (Mather, 1990), and China, followed by eastern North America (Malhi et al., 1999; Houghton, 1995). Large areas of temperate forests are managed for timber (Malhi et al., 1999; Nabuurs et al., 2003; FAO, 2009) either as plantations or managed natural woodlands.

**Table 1 Carbon fluxes in temperate forests**

Source	Forest Type	Age	Carbon Flux (MgC/ha/yr)		
			NPP ("+" = sink)	NEP ("+" = sink)	NEE ("-" = sink)
Barford et al., 2001	1 Oak-dominated hardwood Massachusetts, USA	30 - 100			-2.0
Fahey et al., 2005	1 Northern hardwood New Hampshire, USA	70 - 100	4.83	0 +/- 0.2	
Carrara et al., 2003	1 mixed conif/decid Belgium	70			1.1
As reported in Carrara et al., 2003	1 Deciduous EuroCarboFlux sites	80 - 100			-1.77 to -4.6
As reported in Carrara et al., 2003	1 Coniferous EuroCarboFlux sites	65 - 85			-3.19 to -7.2
Knohl et al., 2003	1 European beech Germany	old growth (250)			-4.9
Gough et al., 2008	1 mixed northern hardwood Michigan, USA	6-90 (ave. 85)		1.5	
Hanson et al., 2003	1 Oak Tennessee, USA	58 - 100	7.29 +/- 0.69	1.87 +/- 0.67	
Malhi et al. 1999	1 Oak-Hickory Tennessee, USA	55			-5.85
Valentini et al. 2000	1 Mixed conifer France	29		4.3	
Wofsy et al., 1993	1 mixed deciduous/hemlock Massachusetts USA	50-70			-3.7 +/- 0.7
Yuan et al., 2008	1 White pine Ontario, Canada	65		1.62	
Yuan et al., 2008	1 Balsam-fir New Brunswick, Canada	27		5.08	
Law et al. 2003	2 Ponderosa pine Oregon, USA	20	2.08	-1.24	
Law et al. 2003	2 Ponderosa pine Oregon, USA	70	4.00	1.18	
Law et al. 2003	2 Ponderosa pine Oregon, USA	100	4.85	1.70	
Law et al. 2003	2 Ponderosa pine Oregon, USA	250	3.32	0.35	
Law et al., 1999, 2000	2 Ponderosa pine Oregon, USA	mixed	4.05	2.66	
Hamilton et al. 2002	4 Loblolly pine North Carolina, USA	15	7.05	4.28	
Maier and Kress, 2000	4 Loblolly pine North Carolina, USA	11	5.0 - 12.35	-1.0 to 7.21	
Jassal et al., 2008	5 Douglas-fir Pacific Northwest, USA	58		3.26	
Yuan et al., 2008	5 Douglas-fir British Columbia, Canada	55		2.73	
Ryan et al., 1996	Radiata pine Australia	20	9.03	2.42	

1 = moist broadleaf and coniferous; 2 = interior coniferous; 3 = montane oak/pine;  
4 = woodland and pineland; 5 = temperate rainforest

Available data show that there have been forest productivity increases across temperate North America, northern Europe, most of central Europe, some parts of southern Europe, and Japan. Most of this sink is occurring in young- to middle-aged stands, with older stands either very small carbon sinks or in some cases, carbon sources, and very young (stand initiation stage) stands acting as carbon sources.

Estimates of carbon stocks in temperate forests range from 98.9 Pg<sup>1</sup> C to 159 Pg C, between 8.6% to 13.8% of the global forest carbon stock (Heath et al., 1993; Dixon et al., 1994; IPCC, 2000). Although temperate forests represent a relatively small portion of forest carbon, most evidence shows that they are currently carbon sinks (Barford et al., 2001; Carrara et al., 2003; Knohl et al., 2003; Wofsy et al., 1993; Table 1), and thus crucial to CO<sub>2</sub> emissions mitigation for the planet. In a review of 31 published studies on national or regional temperate forest productivity, Boisvenue and Running (2006) found that 28 showed increases in above ground net biome productivity during the

<sup>1</sup> Pg (Petagram) = 10<sup>15</sup> grams

20<sup>th</sup> century. They conclude that there have been forest productivity increases across temperate North America, northern Europe, most of central Europe, some parts of southern Europe, and Japan. Most of this sink is occurring in young- to middle-aged stands, with older stands either very small carbon sinks or in some cases, carbon sources, and very young (stand initiation stage) stands acting as carbon sources (Law et al., 2003; Carrara et al, 2003; Mahli et al, 1999; Wofsey et al., 1993). Most temperate forests are currently in “middle age,” from 50 to 100 years old, so growth in this biome should continue to be strong in the near term, but less so in the long-term as these forests age.

## TEMPERATE FORESTS

In general, temperate forests favor the climatic conditions that characterize the humid mid-latitude regions of western and central Europe, eastern North America, and eastern Asia (Archibold, 1994). The northern extent is limited primarily by low winter temperatures (Perry, 1994). Climate in these regions exhibits a marked seasonality; it alternates between warm moist summers and winters mild enough to support broad-leaved angiosperms (Perry, 1994). The growing season lasts 120–250 days and daily temperatures tend to range from -30° to 30°C, with tree photosynthesis occurring between 5° and 25° C (Martin et al., 2001). Precipitation of between 500 and 1,500 mm tends to be either distributed evenly throughout the year, or peaking in summer, with local variation depending on factors such as latitude, topography, and continental position. Exceptionally high precipitation can occur, such as the 9,670 mm recorded on the southwest coast of the South Island of New Zealand (Martin et al., 2001).

Soil is fertile, more often than not enriched with a decaying litter. Soils in Europe are characterized by brown earth – sometimes on calcareous material resembling mollisols. In North America and Asia, alfisols, inceptisols (reflecting the last glaciation), spodosols, and ultisols are common soil types. Soils in the southern hemisphere usually consist of highly podsolized material (spodosols) because of high rainfall and granitic substrates (Martin et al., 2001).

Although the composition of temperate forests is diverse, they can be classified into five major types: *moist broadleaf and coniferous*; *interior coniferous*; *montane oak/pine*; *woodland and pineland*; and *temperate rainforests* (Figure 1).

### Moist broadleaf and coniferous forests

These are mesic, mixed forests with a rich suite of genera, including maple (*Acer*), oak (*Quercus*), hickory (*Carya*), elm (*Ulmus*), linden (*Tilia*), birch (*Betula*), beech (*Fagus*), ash (*Fraxinus*), hemlock (*Tsuga*), and “soft pine” species of the genera *Pinus*. Fire plays a relatively minor role in such forests, except for the “hard pine” dominated forests of sandy coastal plains such as in the U.S. south. They are located in the eastern United States and the Great Lakes shores of southern Canada, central Europe, and northeast Asia (northeast China, Korea and central Japan), dominating the temperate biome. Northeast Asia has the most diverse tree flora with several genera

and families that are not found elsewhere (e.g. *Cunninghamia*, *Cryptomaria*, *Phellodendron*).

Soils classified as ultisols (USDA, 1975) underlie much of this area, particularly in the unglaciated parts of eastern North America, and are generally desirable for cultivation because they are usually relatively fertile (though often stony) and require no irrigation because of precipitation year round. Glaciated regions further north on each continent either have weak soil development (inceptisols) that are often stony and thin to bedrock, or strong organic accumulations and leached upper horizons (spodosols) because colder and high rainfall inhibits decomposition. All these regions are exposed to both seasonal warm and cold air masses, which cause this forest type to have four distinct seasons. Temperatures vary widely from season to season with yearly temperatures averaging about 8°C; precipitation ranges between 750 to 1,500 mm spread fairly evenly throughout the year (Reich and Frelich, 2002).

With extensive clearing for agriculture and settlement, starting well before medieval times in some places, very little of their former extent remains in Europe. What does remain is significantly degraded, or has been afforested with exotic plantations (Nabuurs et al., 2003). In North America, they are almost entirely even-aged second growth, after a period of exploitation and clearing in the 19<sup>th</sup> and early 20<sup>th</sup> centuries. In some places they have nearly reached pre-clearing extent, but in others they are remnants in a sea of urban/suburban development or intensive large scale agriculture of the midwest (Riitters and Coulston, 2005). Forests in Korea and Japan have similar histories to those of the eastern United States. In the last decade China has reversed deforestation with significant recent second growth development and large scale plantation reforestation schemes (FAO, 2009).

### **Interior coniferous forests**

Dry, fire-adapted, in harsh continental climates, interior coniferous forests are often found on andisols (volcanic origin) or mountain inceptisols (from the last glaciation). “Hard pines” (*Pinus*), spruce (*Picea*), fir (*Abies*), poplar, aspen (*Populus*) and larch (*Larix*) predominate. Located in the interior mountain west of the U.S. and Canada, and in Central Asia, these forest types are closely related to interior continental boreal forests. Soils are young, rocky, often skeletal, and exposed to the extremes of cold winters and dry summers. Fire regimes, particularly because of fire suppression in the western U.S., have changed from low intensity, frequent ground fires to high intensity, infrequent stand-replacing fires (McNab and Avers, 1994). Precipitation occurs mostly in winter as snow, with the growing season in spring strongly dependent on snow melt.

### **Montane oak/pine forests**

Pine- (*Pinus*) and oak- (*Quercus*) dominated systems in mountain ranges of Mexico and Central America, the Himalayas, the Mediterranean and Turkey (Central Asia) are fire-adapted and relatively dry. These forests are characterized by hard pines and evergreen oaks (very diverse) on elevation gradients of oak-pine mixtures changing



from dry low to moist high (1,000 m-3,000 m), groundstory fires, and rainfall mostly in winter time, with dry summers. It is a very important region for terraced agriculture, wheat, and corn. Oak leaves are used for fertilizer, fuelwood, and forage for livestock. Soils are alfisols or montane origin inceptisols from the last glaciation. The forest type can be considered a low latitude “hotter” and more droughty variant of the cooler higher latitude interior coniferous forests.

### Woodland and pineland forests

These usually coastal forests are fire-adapted, often open forests in dry, southern temperate climates. They include “hard” pine (*Pinus*) and oak (*Quercus*) in the coastal Mediterranean region, *Acacia-Eucalyptus* savannas of Africa and Australia, and oak-pine woodlands of Mexico. Soils that are generally classified as alfisols (USDA, 1975) predominate. Such soils are more fertile than ultisols but often require partial irrigation because of drier summers. Most forests with alfisols have already been cleared for cultivation, and thus this type is restricted to degraded relics. Rainfall is 500-700 mm per year, occurring in wintertime – summers are dry.

### Temperate rainforests

Mesic, constantly moist, and often extremely productive forests of mountain ranges along coasts. Spruce (*Picea*), hemlock (*Tsuga*), fir (*Abies*), Douglas fir (*Pseudotsuga*) and western cedar (*Thuja*) dominate in the Pacific Northwest; the southern beech (*Nothofagus*) in southwest coastal fringe of Chile and Tierra del Fuego, Argentina; and southern beech, Eucalypts (*Eucalyptus*) and podocarps (*Podocarpus*) in New Zealand and southeast Australia. Spodosols and andisols are the predominant soil types. Andisols are volcanic soils that with high precipitation can be very productive for pasture. Spodosols are acidic soils associated with bedrock geology, predominantly comprised of minerals such as quartz and silica, and are therefore often nutrient poor. Rainfall is > 2,000 mm, sometimes year round, although summers can be drier.

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Most evidence points to a temperate forest biome carbon sink of about 0.2 to 0.4 Pg C/year, with the largest sink occurring in North America and Europe.

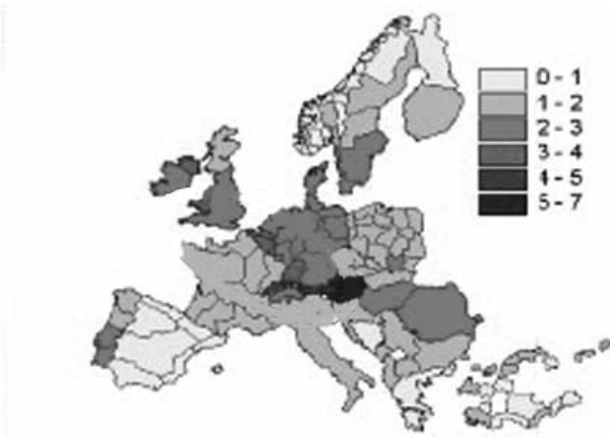
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## TEMPERATE FORESTS AS CARBON SINKS

Most evidence points to a temperate forest biome carbon sink of about 0.2 to 0.4 Pg C/year, with the largest sink occurring in North America and Europe. According to Dixon et al. (1994), in the 1990s, temperate forests in Europe and the continental U.S. were slight carbon sinks (on the order of 0.10 Pg C/year), China a slight source (0.2 Pg C/year), and the total biome a carbon sink of about 0.26 +/- 0.09 Pg C/year, or about 37% of the total net terrestrial carbon uptake (IPCC, 2000), disproportionately higher than their representative area. Luyssaert et al. (2007), in a comprehensive review of global databases of forest ecosystem carbon budgets, conclude that temperate moist broadleaf and coniferous forests have a mean carbon flux (net ecosystem productivity) of about 0.3 – 0.4 Pg C/year, on the upper end of the range of the Dixon et al. (1994) estimates, but very close considering the coarseness of the analysis in both cases.

The European forest carbon sink is estimated variously at 0.068 to 0.7 Pg C/year (Nabuurs et al., 2003; Nabuurs et al., 2008; Figure 2), 0.1 – 0.2 Pg C/year (Janssens et al., 2003) and 0.25 – 0.47 Pg C/year (de Vries et al., 2006). At the low end, this fits well with the temperate forest biome estimates; however, at the high end, which Nabuurs et al. (2003) derived by scaling up eddy covariance flux measurements from experimental sites, it would put the biome number quite a bit higher than other estimates. Hanson et al. (2003) concluded that the reason that net ecosystem exchange (NEE) of CO<sub>2</sub> (from eddy covariance measurements) was higher than net ecosystem productivity (NEP) (difference between photosynthesis and respiration, from biometric analysis) in an upland oak forest in the U.S. is because the allometric relationships used in the biometric analysis do not include the total nonstructural carbohydrate pool (sugars and starches). They found large inter-annual changes in the size of this pool, similar in magnitude to annual increments in fiber or wood. If this holds true for the larger biome, then the higher estimates from flux measurements could be a better reflection of the actual carbon flux from temperate forests (Hanson et al., 2003).

**Figure 2 Net ecosystem production (NEP) in European forests (Mg C/ha)**



Source: Nabuurs, et al., 2008. Hotspots of the European forests carbon cycle. *Forest Ecology and Management* 256, 194-200. Reprinted with permission.

In a review of European carbon flux measurement sites (EuroCarboFlux), Carrara et al. (2003) found that all but one temperate site (a 70 year old Scots pine (*Pinus sylvestris*) plantation in Belgium) were a net carbon sink ranging from 0.2 to 0.7 Mg C/ha/year. European sinks are attributed to both the expansion of forest area and an increase in growth (primarily tree biomass) because of the low average age (60 years) and low average standing volume (Nabuurs et al., 2003). Woodbury et al. (2007) estimated the U.S. forest carbon sink to be 0.1 Pg C in 2005, lower than previous estimates (Birdsey and Heath, 1995), possibly due to improved data. Flux measurements at the Harvard Forest, Massachusetts, USA, show a fairly constant sink of 1.9 to 2.5 Mg C/ha/yr from 1992 to 2000, which is somewhat higher than estimates from biometric data at the same site of 1.6 +/- 0.4 Mg C/ha/yr (Barford et al., 2001).

The temperate forest carbon sink is primarily because of two factors: i) 20<sup>th</sup> century reforestation on former agricultural land in the southern U.S., parts of Europe – and more recently China (FAO, 2009; Smith et al, 2009); and ii) strong growth of secondary 60-80 year old forest stands, particularly in North America, but also in parts of Europe and Asia (Japan, Korea) (FAO, 2006). Recently there appears to be less abandonment of agricultural land and conversion of former cropland to forest plantations in the United States (Smith et al., 2009), and very large reforestation efforts in northeast Asia – particularly China (FAO, 2009). It is unclear how these counteracting shifts in age-classes across the biome will affect the sink status of temperate forests.

### POOLS OF CARBON IN TEMPERATE FORESTS

Temperate moist broadleaf and coniferous forests are without a doubt the most studied forest ecosystems in the world. Less work has been done on the montane pine-oak forests and the woodland and pinelands of coastal dry regions. For this review, carbon pool data was analyzed from 26 published studies, mostly in the United States, but also in Mexico, Canada, Japan, Australia, and Europe (Table 2) to arrive at broad estimates of carbon storage across age classes and forest types. Most of the published literature is from moist broadleaf and coniferous forests, which by far span the largest geographic area of all temperate forest types (Figure 1). Nonetheless, we were able to include several studies from each of the other types in our analysis. Data from the 14 studies from which it was possible to estimate total ecosystem carbon indicate that temperate forests store a vast range of carbon, from a low of about 60 Mg C/ha (young stands) to a high of just over 1,000 Mg C/ha. If we leave out the highest number, which is from an old growth Pacific Northwest, USA rainforest, the high end of the range is around 340 Mg C/ha. Mid-aged (40-80 years) moist broadleaf and coniferous forests contain 100-300 Mg C/ha, about the same as young (50 years) temperate rainforests (210 Mg C/ha). These numbers square fairly well with the biome carbon numbers – a range of 100-300 Mg C/ha equates to 104-312 Pg carbon over 10.4 million km<sup>2</sup>, compared to the IPCC (2000) estimate of 159 Pg C in the temperate forest biome.

However, there is tremendous variability between forest types and age classes, and this carbon stock could easily move to the lower end of the range if disturbance or land use change shifts temperate forests to younger age classes or climates become drier or more seasonal. On the other hand, if temperate forests are managed for longer rotations or more area in old growth reserves, then the carbon stock will shift towards the higher end of the range.

Forest carbon is stored in distinct, but related pools, each of which has a unique response to biotic and abiotic factors that influence carbon uptake, storage, and emission. The major pools in the temperate biome are above ground biomass and soil. Belowground biomass and litter are smaller and less often measured, although both can be considerable (10% or more of total carbon stock), especially in older stands (for example, litter in old growth Ponderosa pine (*Pinus ponderosa*) (Law et al., 2003) or below ground in Douglas fir (*Pseudotsuga menziesii*) (Yuan et al., 2008) and montane oak/pine (Ordóñez et al., 2008).

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Table 2 Carbon stocks in various pools in temperate forests

Source	Forest Type	Stand Age	Carbon Pools (Mg C ha <sup>-1</sup> )					Soil sample depth (cm)
			Above Ground *	Below ground	Litter & CWD	Organic Soil Horizons	Soil	
Barford et al., 2001	1 Oak-dominated hardwood Massachusetts, USA	30 - 100	100					
Fahey et al., 2005	1 Northern hardwood New Hampshire, USA	70 - 100	95	25	13	30	127	20+
Bascietto et al., 2004	1 European beech Germany	70 - 150	132-177					
Edwards et al., 1989	1 Oak/hickory Tennessee, USA	41 - 83	92 - 109			15 - 16	55	100
Fang et al., 2005	1 All - Japan		27.6	6				
Finzi et al., 1998	1 Mixed hardwood/hemlock Connecticut, USA						59 - 75	15
Gough et al., 2008	1 mixed northern hardwood Michigan, USA	6-90 (ave. 85)	76	23	4		80	
Hanson et al., 2003	1 Oak Tennessee, USA	58 - 100	108		4	4	64	100
Harris et al., 1975; Edwards et al., 1989	1 Tulip poplar Tennessee, USA	41 - 83	90 - 96			2 - 9	97 - 125	100
Malhi et al. 1999	1 Oak-Hickory Tennessee, USA	55	79*		7-11	8-27 (incl. roots)	7 - 55	
Morrison 1990	1 Sugar maple Ontario, Canada	old growth	104 - 122*			14 - 16	185 - 202	100
Ruark & Bockheim 1988	1 Quaking aspen Wisconsin, USA	8 - 66	17 - 74*		4 - 8	4 - 9	33 - 65	60
Yuan et al., 2008	1 White pine Ontario, Canada	65	83	17				
Peichl and Arain, 2006	1 White pine Ontario, Canada	15	40	5			34	
Peichl and Arain, 2006	1 White pine Ontario, Canada	30	52	9			30	
Peichl and Arain, 2006	1 White pine Ontario, Canada	65	100	19			37	
Yuan et al., 2008	1 Balsam-fir New Brunswick, Canada	27	78	18				
Law et al. 2003	2 Pinus ponderosa Oregon, USA	20	6	3	12		99	100
Law et al. 2003	2 Pinus ponderosa Oregon, USA	70	53	17	10		76	100
Law et al. 2003	2 Pinus ponderosa Oregon, USA	100	102	33	20		102	100
Law et al. 2003	2 Pinus ponderosa Oregon, USA	250	134	42	14		64	100
Law et al., 1999, 2000	2 Pinus ponderosa Oregon, USA	mixed	98					
deJong et al., 1999	3 montane pine Mexico		127*					
deJong et al., 1999	3 montane pine-oak Mexico		135*					
Ordonez et al., 2008	3 montane pine-oak Mexico		92 - 113	24 - 29	3 - 4		93 - 116	0 - 30
Hamilton et al. 2002	4 Loblolly pine North Carolina, USA	15	51	10				
Hooker & Compton (2003)	4 White pine Rhode Island, USA	10 - 114	8 - 183*			0 - 33	58 - 102	70
Maier and Kress, 2000	4 Loblolly pine North Carolina, USA	11	11 - 22	3 - 7				
Gower et al. 1992	5 Rocky mountain Douglas-fir, New Mexico	50	169*		8	21	11	30
Smithwick et al. 2002	5 Fir-Spruce-Cedar Oregon, USA	150 - 700	120 - 628*			10 - 19	37 - 366	100
Yuan et al. 2008	5 Douglas-fir British Columbia, Canada	55	182	37				
Ryan et al., 1996	Pinus radiata Australia	20	59	12				

\* total living biomass

1 = moist broadleaf and coniferous; 2 = interior coniferous; 3 = montane oak/pine; 4 = woodland and pineland; 5 = temperate rainforest

Most reported research is on either above ground biomass or soil carbon, but not both. Where there are data on both, soil holds between 30% to 60% of total ecosystem carbon in most cases. This is low compared to the IPCC temperate forest biome estimate of 62%. This may be due to differences in how below ground and soil organic layers are classified. Soil carbon was highest in the one old growth sugar maple (*Acer saccharum*) stand, with a range of 184-202 Mg C/ha (Morrison, 1990) compared to 25-130 Mg C/ha for mid-aged moist broadleaf and coniferous forests (Table 2). In temperate rainforests, the carbon budget may be different, as indicated by an old-growth stand with from 65% to 80% of the carbon stored above ground in large trees (Smithwick et al., 2002). Carbon allocation also depends on soil type. Spodosols (infertile/wet-cold) such as those in northern hardwoods contain a greater proportion of carbon in soil (Fahey et al., 2005) versus ultisols (fertile/wet-warm) in more southern hardwood types (Edwards et al., 1989).

### Aboveground biomass

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Successional dynamics and disturbance play an important role in carbon uptake and storage. Younger stands tend to have higher rates of carbon sequestration, as indicated by NEP, than mid- or older-aged stands, although the data are highly variable

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The aboveground pool is the largest pool in temperate systems and is primarily influenced by stand age, with older stands having more above ground carbon than younger stands (Table 2), but also by species composition. European beech (*Fagus sylvatica*) and Douglas fir stands have higher levels of above ground carbon than other types of similar ages (Bascietto et al., 2004; Yuan et al., 2008; Table 2), although in one study the oldest beech stands (150 years) were found to have the lowest carbon density among a chronosequence from 70 to 150 years (Bascietto et al., 2004). Mixed species stands in the moist broadleaf and coniferous forest type tend to have higher carbon density than single species stands (Hanson et al., 2003), possibly due to more heterogeneity of structure and composition, allowing more efficiency in use of light, water, and nutrients. Fir and oak forests within the Mexican montane oak/pine forest type have a higher above ground carbon density than pine or pine/oak forests, although there was little difference in the soil carbon pool (Ordóñez et al., 2008). At the Harvard Forest, Massachusetts, USA, above ground woody increment dominated the carbon uptake in older age classes, even though tree growth rates are relatively slow (Barford et al., 2001).

Successional dynamics and disturbance play an important role in carbon uptake and storage. Younger stands tend to have higher rates of carbon sequestration, as indicated by NEP, than mid- or older-aged stands, although the data are highly variable (Table 1). In a meta analysis of 19 studies from the temperate biome, Pregitzer et al. (2008) found that NPP and NEP both peaked in the 11-30 year age class. This follows the expected pattern of faster growth in young stands, with carbon sequestered via photosynthesis greater than the carbon lost via respiration. In older stands, these two processes tend to be more in balance, making them either neutral or a slight sink (see Chapter 3, this volume, for a detailed analysis of stand dynamics and carbon). Carbon stores, on the other hand, are lowest in young stands and highest in old stands, which can contain up to 2 to 5 times as much total ecosystem carbon as younger stands (Pregitzer et al., 2004; Law et al., 2003; Hooker and Compton, 2003; Peichl and Arian, 2006).

### Belowground biomass

The belowground biomass carbon pool (coarse and fine tree roots and their associated mycorrhizae) is the least studied part of the forest carbon budget, mainly because it is so difficult and labor-intensive to measure the various components of the below ground system. Typically research data indicate that the below ground carbon pool, at around 5% to 10% of total carbon, is much smaller than the above ground (Table 2), however, this may be understated due to estimation methodologies and under-measurement, particularly of fine root biomass (Vogt et al., 1998).

A review of Free Air Carbon dioxide Enrichment (FACE) studies in the United States shows that the contribution of fine roots to the total carbon budget varies greatly by species within broad forest types (conifer, deciduous) (Norby et al., 2005). It was lowest in young trembling aspen (*Populus tremuloides*) (3%) and 20-year old loblolly pine (*Pinus taeda*) plantations (7%), and highest in a 16-year old sweetgum (*Liquidambar styraciflua*) stand (16%). It should be noted that all of these forests are young; it is not known whether the observed differences between forest types would be valid for all age classes.

### Litter and coarse woody debris

The litter pool is made up of dead organic matter (leaves, twigs, debris) on the forest floor that is not completely decomposed and has not yet entered the soil profile. This is the smallest pool, generally less than 10% of total ecosystem carbon.

Litter quantity and quality is a function of species composition, and thus varies between forest types. There can be large interspecific differences in forest floor carbon (Finzi et al., 1998). Carbon in the litter pool has a rapid turnover, compared to most other pools (with the possible exception of fine roots), moving into the atmosphere as respired CO<sub>2</sub> and into the soil in organic carbon compounds. The largest gross fluxes in European forest carbon flux experiments were found to be in the litter (0.316 Pg C/year) and soil organic matter decomposition (0.392 Pg C/year) (Nabuurs et al., 2003).

Deciduous forests receive large inputs of litter in the fall, as trees and understory plants senesce, which decompose slowly during the winter months, and quickly in the growing season. Coniferous forests produce less litter, but it generally decomposes more slowly due to high lignin content, so that the litter carbon pool is not very different than in deciduous forests.

### Soil carbon

Data on mineral soil carbon stocks in temperate forests can only be considered approximations at this time as there is very little research on deep soil carbon. Current estimates are that soils contain at least half the carbon in temperate forests (Figure 3) and possibly as much as two-thirds (IPCC, 2000).

A large study of temperate forest biome soils – based on over 1,000 samples – produced an estimate of 60-139 Mg C/ha (at 0-100 cm. depth), with warm moist forests at the lower end and cool moist forests at the upper end of the range (Post et

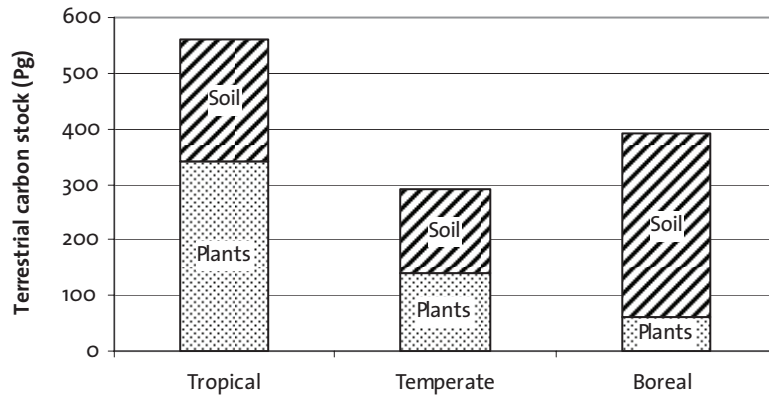
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al., 1982). Extrapolated to 10.4 million km<sup>2</sup> area of temperate forests, this equates to 62.4-144 Pg C soil carbon for the biome. The IPCC (2000) estimate of 100 Pg C is in the middle of this range. Most data on individual sites in our analysis fit well within this biome estimate (Table 2), although the one old growth sugar maple site had much higher soil carbon density (185-202 Mg C/ha) as did the mixed conifer temperate rainforests of Oregon (up to 366 Mg C/ha).

**Figure 3** Distribution of world forest carbon stocks by biome



Source: Data compiled from Vogt et al., 1998; Eswaran et al., 1995; Goodale et al., 2002; Guo and Gifford, 2002.

The vast preponderance of soil carbon measurements are taken in the top 15-100 cm (Table 2). Without data on carbon content of the deep mineral soil, it is difficult to accurately size the soil carbon pool – although glaciated soils that are shallow to bedrock (Gough et al., 2008, for example) are likely more accurate estimates as compared to in-situ weathered soils (as in Hanson et al., 2003). One estimate in an upland oak forest is that the deep mineral soil (1-9 m) contains more carbon (88 Mg C/ha) than the upper 1 m (64 Mg C/ha) (Hanson et al., 2003), which implies a large, relatively stable carbon stock that is not accounted for in most temperate forest carbon budgets.

Carbon enters the soil through intermixing and leaching of decomposed litter, incorporation of earthworm casts and other stable aggregates formed by earthworm activity (Bohlen et al., 2004b), and from fine root turnover. Models by Rasse et al. (2001) predicted that fine root turnover was the single most important source of carbon to beech and Scots pine forests in Belgium. Besides moving carbon into mineral soil, earthworms can also have a negative effect by mixing soil layers and exposing deeper recalcitrant carbon pools to greater mineralization – in areas with exotic invasive earthworms, a more rapidly decomposing litter layer has caused a sharp decline in soil carbon pools in North America (Bohlen et al., 2004a).

As indicated by the data in Post et al. (1982), soil respiration is higher in warm, moist forests than in dry or cool, moist systems. Precipitation, temperature, composition of the microbial community, and nutrient availability are equally

important in determining soil respiration rates (de Deyn et al., 2008). Hence, northern temperate forests, with cooler climates, higher precipitation, and extensive mycorrhizal fungi associations, should have larger, more stable stores of soil carbon than southern temperate forests. It is difficult to determine whether or not this is true because there is so much variability in published data. This variability could be due to differences in measurement methodology and depth at which samples are taken as well as individual species and site factors. For example, significant inter-specific differences were found in soil carbon in a mixed hardwood/conifer forest in Connecticut, USA, particularly in the 7.5-15 cm layer (Finzi et al., 1998).

At the Hubbard Brook Experimental Forest in New Hampshire, USA, researchers assume a stable soil carbon pool in their watershed-scale carbon budget analysis; however, they caution that data are not available to validate this assumption because of the extensive sampling that would be required (Fahey et al., 2005). Several chronosequence studies have found soil carbon to be relatively stable across age-classes with similar land use histories (Law et al., 2003; Peichl and Arain, 2006), although there was a modest increase in soil carbon along a chronosequence of white pine (*Pinus strobus*) stands after agricultural abandonment in Rhode Island, USA (Hooker and Compton, 2003). Soil carbon was found to be stable in an unmanaged mixed pine hardwoods forest in Tennessee, USA (Zhang et al., 2007). However, others have estimated that as much as two-thirds of the sequestered carbon in a 55-year old oak-hickory (*Quercus-Carya*) forest goes into soil organic matter (Malhi et al., 1999), indicating that carbon could be accumulating in the soil, although the authors caution that this is a very rough estimate and should only be used as illustrative of general principles. Much depends on the disturbance and land use history of the stand.

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Several chronosequence studies have found soil carbon to be relatively stable across age-classes with similar land use histories.

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## **BIOTIC DRIVERS OF CARBON STORAGE AND FLUX**

### **Photosynthesis and autotrophic respiration**

The carbon sink/source status of a forest depends on a delicate balance between plant photosynthesis and plant (autotrophic) and decomposer (heterotrophic) respiration. This balance is highly dependent on temperature, moisture, available nutrients, and light, and little is known about how the interactions among these factors influence key environmental variables, such as drought, to affect ecosystem carbon flows (Hanson and Welzin, 2000).

What we do know is very ecosystem-, or even forest stand-, specific. Information comes from individual site studies, with varying land use history and site conditions. Results are highly variable and cannot be meaningfully extrapolated to the biome as a whole. Temperate forest net primary productivity (NPP), a measure of the carbon uptake in photosynthesis minus that lost through autotrophic respiration, has variously been found to correlate with: the combined influence of temperature and precipitation in Catalonia Spain (Martinez et al., 2008); spring snowpack depth, summer temperatures, and the Pacific Decadal Oscillation index in the Pacific



Northwest, USA (Peterson and Peterson, 2001); atmospheric nitrogen deposition in Finland and Russia (Elfving et al., 1996; Ericksson and Karlsson, 1996); and length of the growing season in Austria, Belgium, and the Pacific Northwest, USA, among others (Hasenauer et al., 1999; Carrara et al., 2003; Peterson and Peterson, 2001).

At the Harvard Forest, Massachusetts, USA, annual CO<sub>2</sub> exchange was found to be particularly sensitive to length of growing season, summer cloud cover, winter snow depth, and drought in summer. The first two regulate photosynthesis, and the latter two affect decomposition and heterotrophic respiration (Goulden et al, 1996). Changes in any of these factors would either increase or decrease the amount of carbon sequestered and stored in the ecosystem. For example, microbial decomposition may be limited by freezing (Goulden et al, 1996), so colder winters, heavier snow packs, or earlier fall freezes will decrease heterotrophic respiration. Higher spring temperatures will bring about earlier leaf-out, and a longer period of spring carbon uptake.

Many studies have looked at ecosystem responses to temperature variations, and the responses differ. There is an optimal temperature (50- 250 C) for photosynthesis in trees (Malhi et al., 1999), and in temperate climates it generally occurs from mid-to-late spring until mid-summer. Higher spring temperatures will enhance carbon uptake, as happened at Asian flux sites during a spring high temperature anomaly in 2002 (Saigusa et al., 2008), but higher summer temperatures may increase ecosystem respiration because of a direct effect on soil temperature (Yuan et al., 2008). Using MODIS “greenness” data, Potter et al. (2007) found that U.S. forests were largely a sink in 2001, 2003, and 2004, but a source in 2002 when the annual mean temperature was above average in the northeast regions.

However, temperature (or any environmental variable) alone cannot explain temperate forest carbon flux dynamics. A few examples bear this out. Hanson et al. (2003) found no relationship of NEP to mean annual temperature in a review of 7 studies of U.S. temperate deciduous forests; there was a positive, but not strong, relationship to precipitation. Whereas ecosystem carbon storage was found to increase with altitude in the Great Smokey Mountains (Tennessee, U.S.), attributed to decreased respiration at higher elevation due to lower temperatures and higher precipitation (Zhang et al., 2007). Net flux of carbon dioxide, (NEE) at the Harvard Forest, Massachusetts, USA, has been observed to respond quickly to short term changes in climatic conditions such as temperature, precipitation, and snow cover, attributed to changes in rates of decomposition (Barford et al., 2001). Although there was very little response in levels of photosynthesis to changes in environmental variables from summer to summer, there were large shifts observed in annual NEE resulting from brief anomalies in temperature during April and May (Goulden et al. 1996). In a Michigan, USA, northern hardwoods site, high year-to-year fluctuations in carbon storage were observed to correlate with variations in air temperature, whereas respiratory losses were correlated with winter temperatures (Gough et al., 2008). The largest effect was found to be from a combination of high temperature and reduced radiation, which lowered mean annual carbon storage by 28% (Gough et al, 2008).

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Temperature (or any environmental variable) alone cannot explain temperate forest carbon flux dynamics.

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### Length of growing season

Temperate forests are strongly seasonal, with a well-defined growing season that depends primarily on light (day length) and temperature. This is probably the most important determinant of temperate forest productivity, or carbon uptake. For example, at one EuroCarboFlux site in Belgium, it has been observed that NEE is highly correlated with the length of the growing season – the forest is a carbon sink in the growing season, between May and August, and a source in the dormant season, from September to April (Carrara et al., 2003).

Most carbon uptake occurs in the spring and early summer, so higher temperatures earlier in the spring will generally increase annual productivity. Other critical factors are moisture, particularly towards the end of the growing season, the timing of the last frost in the spring and first frost in the fall, and summer temperature (higher temperature in summer causes high evapotranspiration, so plants will compensate by closing stomata).

There is evidence of a longer growing season in North America and Europe over the last 50-100 years. Data from the Long Term Ecological Research site at Hubbard Brook, New Hampshire, USA, indicate that the timing of spring melt has advanced from 10-12 days, and green canopy duration has increased by about 10 days since 1958, with significant trends towards an earlier spring (as evidenced by sugar maple leaf-out) (Vadeboncoeur et al., 2006; Richardson et al., 2006). In France and Switzerland, the onset of phenology has advanced considerably in response to spring temperature increases over the last 100 years (Schliep et al., 2008). Higher net carbon uptake in European forests in the spring of 2007, a year of record warm spring temperatures, was attributed to phenological responses to temperature (early bud break in deciduous trees and early release from winter dormancy in conifers) (Delpierre et al., 2009).

The net effects of a longer growing season on carbon sequestration are unclear, and may be confounded by other climate variables such as drought. Satellite observations (combined normalized difference vegetation index data set and climate data) suggest that in mid- to high-latitudes, decreases in carbon uptake during hotter and drier summers offset increased uptake in the spring, thereby reducing or even eliminating the positive benefits of a longer growing season (Angert et al., 2005). However, site-specific ecosystem flux data do not bear this out. For example, carbon flux measurements over an evergreen Mediterranean forest in southern France suggest that increased severity of summer drought did not negatively affect the carbon budget of the ecosystem, and that the annual variability in NEP cannot be fully explained by drought intensity, but is significantly linked with the length of the growing season (Allard et al., 2008). And at two AmeriFlux sites, earlier growing season onset resulted in an increase in net ecosystem productivity both in the spring and over the entire growing season, which the authors suggest could be a result of accelerated nitrogen cycling rates later in the growing season (Richardson et al., 2009).

### Heterotrophic respiration and decomposition

Decomposition of organic matter, a form of respiration, emits carbon dioxide, a “flux,” from the ecosystem to the atmosphere, primarily from short-lived carbon

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Temperate forests are strongly seasonal, with a well-defined growing season that depends primarily on light (day length) and temperature. This is probably the most important determinant of temperate forest productivity, or carbon uptake.

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pools such as soil organic carbon and fine roots (Trumbore, 2000). Decomposition and respiration rates in temperate forests depend strongly on soil temperature (Savage et al., 2009; Zhu et al., 2009; Jassal et al., 2007) and moisture (Jassal et al., 2007; Cisneros-Dozal et al., 2007), but also vary with litter quality (Fissore et al., 2009) and nutrient availability (Fahey et al., 2005). Fissore et al. (2009) found that mean residence time of active (acid soluble) soil organic carbon decreased strongly with increasing temperature in 26 deciduous and coniferous forest sites along a 22° C temperature gradient in North America, confirming a positive temperature influence on heterotrophic respiration across forest types. At the Harvard Forest, Massachusetts, USA, Borken et al. (2006) found that experimental moisture stress caused a decrease in heterotrophic respiration that was not wholly counteracted by increased respiration from natural precipitation levels the following season, resulting in at least a short term net carbon sink.

Fine root respiration was found to vary with temperature in soils at Hubbard Brook, New Hampshire, USA, but was much higher for roots in the forest floor than in the soil at all temperatures, attributed to higher nutrient concentration (particularly nitrogen) in root tissues in the forest floor (litter layer) (Fahey et al., 2005).

According to Dalal and Allen (2008), elevated CO<sub>2</sub> increases soil respiration rate, possibly due to the enhanced rate of fine root turnover. From the limited evidence on soil respiration and climate variables in temperate forests, it appears that higher temperatures and increased precipitation will increase respiration and hence carbon emissions, whereas lower temperatures and drought conditions will decrease respiration and lower CO<sub>2</sub> emissions.

## **DISTURBANCE AND ABIOTIC DRIVERS OF CARBON STORAGE AND FLUX**

The net carbon accumulation in forests is heavily dependent on the time elapsed since disturbance (Pregitzer et al., 2004; Peichl and Arain, 2006; Hooker and Compton, 2003), because disturbance creates biogeochemical changes (light, temperature, moisture, nutrients) that affect both growth and respiration (Pregitzer et al., 2004). Fire, drought, windstorms and ice storms, insects and pathogens, nitrogen and other pollutants, and forest management and land use change are the primary natural and human disturbances affecting temperate forests.

### **Fire**

Although fire plays a minor role in moist broadleaf and coniferous forests, it is a part of the natural disturbance regime in the fire-adapted interior coniferous, montane oak/pine, and woodland and pineland forests. In the United States alone, an average of 3.3 million hectares burn in wildland fires<sup>2</sup> each year, mostly in the west, but also in the southeast and midwest (National Interagency Fire Center, 2009). The area impacted by wildfires has increased significantly in the last ten years (Figure 4). According to the reconstruction of fire history by Mouillot and Field (2005), this is still much lower than in the first decade of the 20th century, when they estimate that fires burned close to 30 million hectares per year. Their analysis shows fires increasing

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In the United States alone, an average of 3.3 million hectares burn in wildland fires each year, mostly in the west, but also in the southeast and midwest.

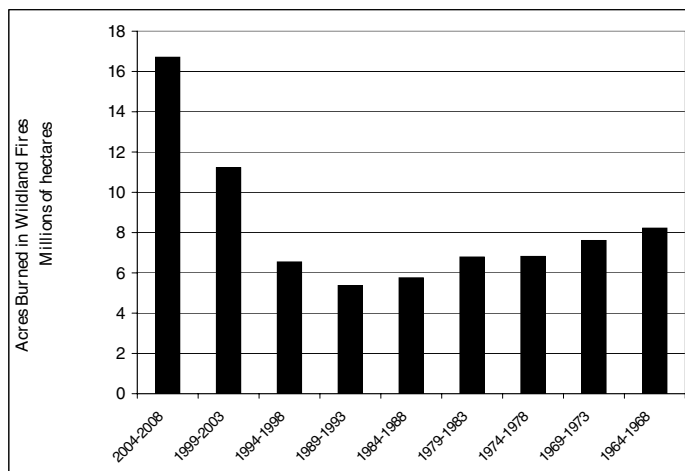
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<sup>2</sup> Includes all wildland fires on grasslands and forests.

in Europe towards the end of the 20<sup>th</sup> century; however, at around 0.5 million hectares per year, the area burned is much smaller than in North America.

Dale et al. (2001) predict a 25-50% increase in burned area throughout the United States over the next 100 years. Understanding the effects of fires on landscape carbon storage over both short-and long-term temporal scales is critical to predicting future changes in both the regional and global carbon budgets (Kasischke et al., 1995). The alteration of ecosystem carbon balance (net ecosystem production, or NEP) varies with time between fires and fire intensity. During a fire, carbon is released to the atmosphere through combustion, creating an immediate CO<sub>2</sub> emission and reduced net primary production (NPP) due to tree mortality. If a stand replaces itself, then the net carbon balance may be zero over a long fire cycle (Kashian et al., 2006). However, net carbon loss to the atmosphere due to increased decomposition and reduced biomass can persist for over a century (Crutzen and Goldammer, 1993).

**Figure 4 Wildland fires in the United States.** Total hectares burned in 5-year periods from 1964-2008.



Source: Data derived from National Interagency Fire Center, 2009. Wildland Fire Statistics 1960 – 2008. National Interagency Coordination Center, Boise, Idaho. [http://www.nifc.gov/fire\\_info/fires\\_acres.htm](http://www.nifc.gov/fire_info/fires_acres.htm)

Short-term effects of fires (from a few years to decades) are important for predicting the Earth's carbon balance over the next century because greater fire frequency, extent, or severity will release current carbon stores through combustion and result in a negative NEP (Kashian et al., 2006). If burned area significantly increases over the next century, these short-term effects will likely influence atmospheric CO<sub>2</sub> concentration (Dale et al., 2001). Short-term effects of fire on carbon storage are regulated by the amount of carbon lost in combustion (Tinker and Knight, 2000; Litton et al., 2004), by the rate and amount of regeneration (Kashian et al., 2004; Litton et al., 2004), and by changes in decomposition rates from altered soil conditions and increased woody debris left by the fire (Auclair and Carter, 1993; Kurz and Apps, 1999).

Long-term effects of fire (over many centuries) on ecosystem carbon balance are regulated by processes that control post-fire regeneration and by fire frequency. If the post-fire stand has poor or no regeneration, forest growth will not replace the carbon lost through combustion and decomposition, and the net carbon storage over a fire cycle will decrease (Kashian et al, 2006). Changing fire frequency will also affect the net carbon storage because the amount of carbon stored in a stand, and the rates of photosynthesis and decomposition, vary with stand age (Kasischke, 2000). It is also important to note that more frequent fires will promote a higher proportion of young forests, and these forests tend to store less carbon than older stands because they contain less biomass, even though their rates of production tend to be higher (Ryan et al., 1997). Thus, if changing climate alters the frequency and intensity of fires, re-vegetation and patterns of carbon storage will likely be affected, particularly in interior coniferous forests.

Although fire was not historically as severe a disturbance in moist broadleaf and coniferous forests as in woodland and pinelands and interior coniferous forests, it nonetheless has played an important historical role (Pyne, 1982). In eastern North America, Native American burning and lightning resulted in relatively frequent fire in temperate mixed oak and pitch pine (*Pinus rigida*) forests. Fire was also important to drier regions, such as near the prairie–woodland border. These fires had a tremendous impact on the composition and age structure of the forest, since certain species have adaptations, such as thick bark, ability to sprout, and rapid post-fire colonization, that enable them to thrive under such conditions (Reich et al., 1990; Abrams, 1992; Kruger and Reich, 1997; Peterson and Reich, 2001). Fire frequency regulated the balance between late successional species such as sugar maple, beech (*Fagus grandifolia*), and linden (*Tilia* spp.), and shade-intolerant early successional species such as oak and aspen (*Populus* spp.), which were abundant along the prairie-forest border and areas with sandy soil where fires were most frequent (Grimm, 1984; Abrams, 1992).

With increasing development following European settlement and expansion, fires in U.S. moist broadleaf and coniferous forests became much less frequent. This was due to cessation of intentional burning, direct suppression of fires, and land use changes that disrupted the contiguity of burnable vegetation across the landscape. Hence, these forests have gradually become increasingly dominated by shade tolerant species such as maple (*Acer* spp.), beech, ash (*Fraxinus* spp.), and linden, with decreased abundance of oaks (Crow, 1988; Abrams, 1998). In the absence of fire, oaks do not establish well in either shaded understories or sunlit openings, because they are neither shade tolerant nor fast growing (Reich et al., 1990; Abrams, 1992; Kruger and Reich, 1997). Hence, a major change in temperate deciduous forests of North America has resulted from the ascendancy of fire-intolerant species to a dominant position in these regions. Fire suppression has likely had similar effects in Europe and Asia, but the longer time since active fire regimes makes it more difficult to be specific about the changes that have occurred (Reich and Frelich, 2002).

### Drought

Water availability controls tree growth, tree species distribution, and forest composition more than any other perennial factors (Hinckley et al., 1981). Global

circulation models predict that increasing concentrations of atmospheric CO<sub>2</sub> will increase the severity and frequency of drought in regions where temperate forests are found (Pastor and Post, 1988; Dale et al., 2001). However, there is a great deal of uncertainty how drought will affect carbon cycles.

Elevated CO<sub>2</sub> generally increases instantaneous water-use efficiency in tree seedlings (Jarvis, 1989), but may have negative impacts on other physiological processes. For instance, stomatal closure can occur during a leaf water deficit or by a high internal CO<sub>2</sub> concentration (Hinckley et al., 1981), which may result in an increased resistance to CO<sub>2</sub> uptake (Jarvis, 1989). Tschaplinski et al. (1995) also found that drought may slow the growth rate and alter the gas exchange of several tree species (including maples) growing in an elevated CO<sub>2</sub> atmosphere.

Given a slowed growth rate or altered gas exchange, it is likely that drought may have a negative impact on regional carbon budgets in the short-term (i.e. during the period of the drought or for several years following a drought event), but it is unlikely that it will affect the carbon cycle in the long-term unless there is substantial tree mortality as a result of the drought event. Beerling et al. (1996) note that there will be a greater tendency for trees to show greater drought tolerance in the future and thus, drought may have little consequence on NEP.

### Wind and ice

Perhaps the most important abiotic disturbance regime in temperate forests is wind and ice, creating a mosaic of gaps and gap sizes that drives successional processes across the landscape (Dale et al., 2001; Nagel and Svoboda, 2008). Moist broadleaf and coniferous forests are heavily impacted by wind disturbance, including tornadoes and thunderstorm downbursts in central North America and western Europe and severe extra-tropical low-pressure systems (cyclones and hurricanes) along the eastern Atlantic coast (Dale et al., 2001; Reich and Frelich, 2002; Degen et al., 2005; Nagel et al., 2008).

Most windstorms are small- to intermediate-scale events, resulting in gap formation or more frequently, gap expansion (Worrall et al., 2005), thereby either releasing advance regeneration (accelerating succession) (Webb and Scanga, 2001; Uriarte and Papiak, 2007) or creating conditions for disturbance specialist understory plants to take over the gap (Palmer et al., 2000). Although small-to-intermediate-scale, low intensity windstorms can change the successional patterns and species composition of forest stands (Hanson and Lorimer, 2007; Papaik and Canham, 2006; Degen et al., 2005), they may have little impact on total carbon stocks at the landscape scale, particularly if the downed trees and branches are left on the ground. For example, 23 years after an intermediate windstorm in an old growth beech-fir forest in Slovenia, Nagel et al. (2006) found that, although the basal area of living trees was lower, the basal area of downed logs was higher in areas affected by windthrow.

Hurricanes often create patches of disturbance of intermediate severity across the landscape (McNab et al., 2004; Busing et al., 2009), although intense storms, such as Hurricane Rita along the Gulf Coast in 2005, leave wide swaths of forest damage (Juarez et al., 2008). Carbon moves quickly from the living biomass pool to the dead and downed wood pool (Uriarte and Papaik, 2007; Busing et al., 2009), reducing NPP

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and increasing respiration from decomposition, with a net loss of carbon that can last for decades (Fahey et al., 2005; Busing et al., 2009; McNulty, 2002). Frequent storms have been shown to depress carbon stocks in southern New England, USA. Maturing second growth hardwood forests exhibit a decrease in carbon (living and dead above ground biomass) across a hurricane severity gradient from south (more severe) to north (less severe) (Uriarte and Papiak, 2007). The authors suggest that in the southernmost areas of New England, storm-free periods were never long enough for the forest stands to reach peak biomass.

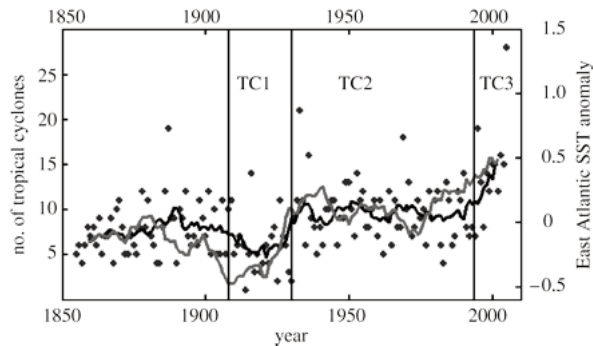
The frequency of such stand-leveling winds is expected to increase under a warmer climate, so that fewer stands would reach old-growth stages of development. Thus there would be a decrease in overall carbon sequestration in regions experiencing severe wind storms (Uriarte and Papiak, 2007). Holland and Webster (2006) looked at 100-year tropical cyclone activity in the eastern Atlantic and concluded that, over the 20<sup>th</sup> century, increased storm frequency is related to rises in sea surface temperature; thus, the recent upsurge in tropical cyclones and hurricanes (Figure 5) is due in part to global warming. From 1995-2007, there were an average of 15 major tropical cyclones (including 8 hurricanes) per year, compared to an average of 9 (5 hurricanes) during the period 1931- 1994 (Holland and Webster, 2007).

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In the southernmost areas of New England, storm-free periods were never long enough for the forest stands to reach peak biomass.

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**Figure 5 Tropical cyclone occurrence** (dots indicate annual totals and the black line is a 9-year running mean) **in the North Atlantic together with East Atlantic sea surface temperature (SST) anomalies for the hurricane season** (grey line) **from 1855 to 2005.**



Source: Holland, G. J., Webster, P. J., 2007. Heightened tropical cyclone activity in the North Atlantic: natural variability or climate trend? *Philosophical Transactions of the Royal Society a-Mathematical Physical and Engineering Sciences* 365, 2695-2716. Reprinted with permission.

Ice storms are common in moist broadleaf and coniferous forest regions (Goodnow et al., 2008; Changnon, 2008), although catastrophic ice storms are rare (Bragg et al., 2003). Injury to trees is widely variable, from minor branch breakage to mortality, and depends on the storm severity, species, and site conditions (Bragg et al., 2003; McCarthy et al., 2006, Boyce et al, 2003). Ice storm damage on the Duke Forest, North Carolina, USA resulted in a transfer of carbon from the living to detrital pools equivalent to 30% of the net ecosystem carbon exchange of the system, with conifers twice as likely to be killed as deciduous trees (McCarthy et al., 2006).

Thinned stands had a three-fold increase in carbon transfer to the detritus pool as compared to unthinned stands. Under elevated CO<sub>2</sub> conditions of the Free Air Carbon dioxide Enrichment (FACE) experimental plots, carbon transfer was significantly less than in the control plots, suggesting that forests might be less susceptible to ice storm damage in a higher atmospheric CO<sub>2</sub> environment.

## Insects

Conjectures about the effects of climate change on insect populations have been somewhat general to date. It is assumed that disturbance intensity will change across a latitudinal gradient as insect populations extend their ranges to higher latitudes and elevations as temperatures rise, with temperate tree species encountering new non-native insects which migrate much more quickly than trees (Williams and Liebhold, 1995; Dale et al., 2001). Such is the case with the mountain pine beetle (*Dendroctonus ponderosae*), which is causing widespread mortality in northwestern North America well beyond its historical range (Kurz et al., 2009). Increased over-wintering survival and higher population growth rates may become more common for insect pests of temperate trees. As such, it is important to understand the impacts that larger pest populations – particularly defoliating insects – will have on forests. Although we provide a few examples here, it must be noted that little literature exists that examines the impacts of insect defoliation on the carbon budget in temperate ecosystems.

Large-scale insect infestations can cause high mortality, leading to long-lasting decreases in ecosystem biomass (Knebel et al., 2007). Kurz et al. (2009) predict that the mountain pine beetle outbreak in British Columbia, Canada, will change the 374,000 km<sup>2</sup> affected area from a small carbon sink to a large carbon source throughout the next decade. Great spruce bark beetle (*Dendroctonus micans* Kug.) outbreaks, interacting with climate stress (cold winters and dry summers) led to forest dieback over 10-15 years in Norway spruce (*Picea abies*) plantations in France (Rolland and Lemperiere, 2004). Invasive exotic species, such as the European gypsy moth (*Lymantra dispar*) and the emerald ash borer (*Agrilus planipennis*) can severely impact temperate forests in the United States, causing widespread defoliation and/or mortality.

Even major defoliation events in deciduous forests may have only negligible effects on NEP, however: after heavy defoliation from a hurricane (similar in effect to insect defoliation) in Florida, USA, the decline in GPP was offset by a concurrent decline in ecosystem respiration (Li et al., 2007). Defoliation, acting with other environmental variables, can also affect nutrient cycling, because large fluxes of organic matter move from one pool (live biomass) to another (detritis and soil organic matter), changing rates of photosynthesis, decomposition, and critical biochemical parameters such as C:N ratios. Two examples from vastly different ecosystems bear this out. Severe defoliation events, combined with recovery from extreme drought, in the 1960s resulted in dissolved inorganic nitrogen losses from the Hubbard Brook Watershed, in New Hampshire, USA, (Aber et al., 2002); and heavy infestations of bark beetle in Ponderosa pine forests in Arizona, USA, did not alter soil respiration rates, but altered nitrogen cycling throughout the growing season, lowering net nitrification rates (Morehouse et al., 2008).

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Disturbance intensity will change across a latitudinal gradient as insect populations extend their ranges to higher latitudes and elevations as temperatures rise, with temperate tree species encountering new non-native insects which migrate much more quickly than trees.

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Increased mortality will more likely result in the release of carbon through decomposition and lower amounts of carbon in living biomass, reducing net carbon sequestration on sites with heavy mortality. Repeated defoliation or attacks will only exacerbate this effect and will likely have negative impacts on the regional or global carbon budget. Furthermore, the rate of recovery or presence/absence of regeneration will also determine the amount of carbon being sequestered within a stand that has been heavily defoliated.

## Nitrogen deposition and ozone pollution

### *Nitrogen*

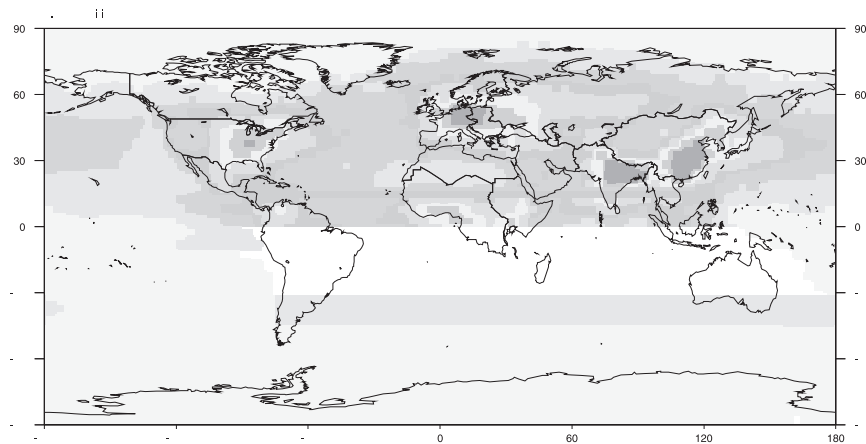
Atmospheric pollution, primarily in the form of nitrogen oxides (NO<sub>x</sub>) emitted from burning fossil fuels, and ozone (O<sub>3</sub>), formed in the atmosphere through the interaction of nitric oxide (NO), sunlight, and hydrocarbons, are chronic stressors in temperate forest regions (Bouwman et al., 2002; Felzer et al., 2007; Figure 6). NO<sub>x</sub> disassociates in the soil solution as hydrogen ions (H<sup>+</sup>) and nitrate (NO<sub>3</sub><sup>-</sup>); the resulting increase in soil pH causes nutrient cations such as calcium (Ca<sup>2+</sup>) and magnesium (Mg<sup>2+</sup>) to leach from the soil, and mobilizes toxic cations such as aluminum (Al<sup>3+</sup>) (Likens and Borman, 1995; Driscoll et al., 2001, Puhe and Ulrich, 2001). Because most temperate forests are thought to be nitrogen-limited, nitrogen deposition may also act as a growth stimulant (fertilizer effect). On balance, however, the evidence regarding nitrogen deposition effects on carbon sequestration is mixed.

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**Figure 6** Global atmospheric nitrogen deposition patterns



Source: Bouwman, A. F., Van Vuuren, D. P., Derwent, R. G., Posch, M., 2002. A global analysis of acidification and eutrophication of terrestrial ecosystems. *Water Air and Soil Pollution* 141, 349-382. Reprinted with permission.

Under current ambient levels, nitrogen deposition is most likely enhancing carbon sequestration, as indicated by data from experimental sites in Europe and the United States. Strong correlations were found between canopy nitrogen levels and canopy-level photosynthetic capacity (CO<sub>2</sub> absorption capacity) across 11 AmeriFlux sites

(Ollinger et al., 2008). It is estimated from EuroFlux data that 10% of net carbon sequestration in Europe is attributed to nitrogen deposition (DeVries et al., 2006). And CO<sub>2</sub> enrichment effects were amplified by high levels of soil nitrogen availability at one FACE site in the southeast USA (Norby et al., 2005).

Nitrogen acts within a complex of stressors including climate change, drought, insects, diseases, and other air pollutants; therefore, efforts to understand the effect of nitrogen fertilization have to be made in the context of these other factors. Thus far, most data come from nitrogen addition experiments where the effects of other factors are intrinsically assumed to be consistent between experimental plots and control plots. The results are inconclusive.

Short-term (one growing season) nitrogen fertilization experiments have produced a decrease in CO<sub>2</sub> emissions, primarily due to decreased soil respiration, in black cherry (*Prunus serotina*) stands (Bowden et al., 2000) and a large increase in NEP in Douglas fir stands (Jassal et al., 2008). Strong responses to nitrogen additions may only be a short-term ecosystem response, however. One-to-three year studies have shown either mixed results (Waldrop et al., 2004) or no detectable change in biomass (Nadelhoffer et al., 1999). Analysis of long-term chronic nitrogen addition experiments in Europe and North America indicate no discernable trend in effects on ecosystem-level carbon sequestration (LeBauer and Treseder, 2008; Evans et al., 2008; Pregitzer et al., 2008; and Bauer et al., 2004). This leads to the tentative conclusion that under chronic nitrogen deposition, temperate forests may no longer be nitrogen limited, also supported by the fact that at several of these sites nitrogen is being exported as nitrate (NO<sub>2</sub>) leaching.

### Ozone

Unlike nitrogen, ozone (O<sub>3</sub>) has no known “positive” effects on forests. High levels of ozone cause foliar injury and consequent growth reduction, particularly in conifers, and all other things being equal, carbon sequestration is expected to be lower in forests with high ozone levels (Augustaitis and Bytnerowicz, 2008). Ozone is highest in areas with high levels of both sunlight and fossil fuel emissions. This includes most of the temperate forest biome (southwestern and eastern United States, eastern Europe, the Mediterranean, western Asia, and northeastern China (Felzer et al., 2007). It is projected that 50% of northern hemisphere forests will be affected by toxic levels of ozone by 2100 (Fowler et al. 1999).

Ambient ozone levels have been associated with growth reduction in mature southern pines, particularly loblolly pine (Felzer et al. 2007). In Europe, ozone has been implicated in growth reductions of Aleppo pine (*Pinus halepensis*) in the Mediterranean basin, Swiss stone pine (*Pinus cembra*) in the timberline ecotone of the European mountains (Richardson et al. 2007), and Scots pine in central Europe (Augustaitis and Bytnerowicz, 2008). Several pine species in Mexico show ozone-induced damage similar to pines in the western United States (Richardson et al. 2007). In the Great Smokey Mountain National Park, USA, ozone stress is thought to be dampening the potential CO<sub>2</sub> fertilization effect, with carbon stocks increasing only slightly between 1971 and 2001 (Zhang et al., 2007).

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All other things being equal, carbon sequestration is expected to be lower in forests with high ozone levels

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### Forest management and land use

Almost all temperate forests have been severely impacted by human use. In Europe and North America, less than 1% of all forests remain in undisturbed state, free of logging, grazing, deforestation or other intensive use (Reich and Frelich, 2002). The largest direct impacts on temperate forests stem from conversion to other land uses. Pre-industrial forest extent is uncertain, but conversion to other types of land use has been profound. In Europe, massive deforestation occurred centuries ago. Native forests have been all but eliminated in some countries, such as Ireland and Britain (Reich and Frelich, 2002). In eastern North America, deforestation migrated westward with agricultural settlement from the 1600s to the mid 1800s. For instance, states originally almost entirely forested, such as Vermont, were more than 80% deforested and converted to agriculture in the 19<sup>th</sup> century. However, many temperate zone economies have shifted from rural agricultural to urban manufacturing and technologically-driven economies. This has led to large-scale agricultural abandonment and the reversion of agricultural lands back to forests. Recent studies indicate an increase in forest area and volumes in both North America and Europe (Kauppi et al., 1992; Houghton, 1995).

Management of forest for timber creates a cyclical pattern of carbon release and sequestration, and intensively managed stands store less carbon than unmanaged forests (Carrara et al., 2003; Gough et al., 2008; Ordóñez et al., 2008; Woodbury et al., 2006). (See Chapter 10, this volume, for a detailed discussion of managing temperate forests for carbon.)

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### THE FUTURE OF TEMPERATE FORESTS AS CARBON RESERVOIRS: CLIMATE CHANGE IMPACTS

Although remaining intact temperate forests continue to be fragmented by development, particularly in North America (Wickham et al., 2008), there is no large-scale deforestation at present nor is there likely to be in the future. Forest cover should remain stable because of conservation efforts in the United States, Japan, South Korea, and Europe, and also because reforestation should balance out the loss of forest cover from development and suburban sprawl. Former agricultural lands continue to be planted with pine in the U.S. south (Smith et al., 2009), forest area is expanding in parts of Europe, particularly Spain and Italy, and recently there have been extensive reforestation efforts with exotic plantations in China (FAO, 2009).

The future of the temperate forest biome as a carbon reservoir and atmospheric CO<sub>2</sub> sink rests mainly on its productivity and resilience in the face of changing disturbance regimes in the context of rising atmospheric CO<sub>2</sub>. The small “sink” status (0.2-0.4 Pg C/year) of temperate forests could easily change to a “source” status if the balance between photosynthesis and respiration shifts even slightly. Predictions are that temperatures in temperate regions will increase (IPCC, 2007); warming in Europe and North America is likely to be largest in the winter, although the Mediterranean and southeastern U.S. are likely to see largest temperature increases in

the summer. Generally this would mean longer growing seasons. Longer and more intense summer heat waves are predicted for East Asia, along with increased precipitation.

There is evidence of increasing productivity in temperate forests as climate has warmed in the last ~50 years; however, this is confounded by successional dynamics and environmental variables. The atmospheric system has not only experienced changes in temperature, precipitation, and radiation, but also in CO<sub>2</sub> concentration and pollutants, between 1950 and 2005 (Keeling et al., 1995; Innes and Peterson, 2001). Current global atmospheric CO<sub>2</sub> is approximately 380 ppm, an increase of about 65 ppm since the 1950s (Keeling and Whorf, 2002). How forests will respond to rising levels of CO<sub>2</sub> in the long term is still uncertain, but the present overall response is positive.

What we know about rising levels of atmospheric CO<sub>2</sub> and forest carbon sequestration comes from a few experimental CO<sub>2</sub> enrichment studies (FACE) in the United States and Europe. A median increase of 23% in net primary production has been recorded across sites exposed to elevated CO<sub>2</sub> (550 ppm) in comparison to control sites (370 ppm) over 1 to 6 years of FACE experiments (Norby and Luo, 2004). In these fast growing, early successional stands, changes in NPP are related to increased atmospheric CO<sub>2</sub> effects on light energy; increased light absorption in stands with a lower leaf area index, and increased light use efficiency in those with a higher leaf area index.

Nowak et al. (2004) tested several early hypotheses on the response of ecosystems to elevated CO<sub>2</sub> using results from FACE experiments in forests and grasslands across North America and Europe. Among these hypotheses were: i) that acclimatization of photosynthesis would occur most prevalently where nitrogen is limiting; ii) that productivity response would be greater in drier ecosystems and in drier years for more humid ecosystems (resource-based response models); and iii) that non-woody functional groups should be more responsive than woody plants (plant functional-type response model). As expected, leaf CO<sub>2</sub> assimilation and ecosystem primary production increased across all species. The primary production observations, however, are mixed and are overall less than the hypothesized 20%. Down-regulation of photosynthesis happened in a number of FACE experiments, but not in all species and not consistently in species among sites. The hypothesis about differing responses depending on site water levels was not well supported, but the predicted increase in productivity enhancement with nitrogen availability was. Nowak et al. (2004) concluded that there was no consistent evidence for either the resource-based or the plant functional-type response model to CO<sub>2</sub>.

Wittig et al. (2005) evaluated GPP of fast-growing *Populus* species (three years from establishment to canopy closure) in response to elevated CO<sub>2</sub> and found that GPP increased dramatically in the first year but markedly less so in the subsequent years. Hättenschwiler and Körner (2003) similarly found accelerated growth in trees over a 30-year period of elevated CO<sub>2</sub> exposure, with most of the accelerated growth happening at young stages of development. In their 2005 analysis based on FACE data, Körner et al. (2005) found an immediate and sustained enhancement of carbon

flux in mature temperate forest trees but, contrary to expectations, found no overall stimulation of growth or litter production after four years; hence, forests seem to be “pumping” carbon through faster with no net gain in biomass (NEP). These findings suggest differing responses of trees at different developmental stages.

Further confounding any simplistic notions of the interactions between atmospheric CO<sub>2</sub> and carbon sequestration in forests are observations about interaction with other environmental variables such as drought. Drought stress is expected to outweigh CO<sub>2</sub> enhanced growth in the southern range of Scots pine, based on findings by Martinez et al. (2008) that summer temperature and water availability have been the main climatic drivers of growth over the past 80 years. Hättenschwiler and Körner (2003) suggest, however, that trees exposed to higher CO<sub>2</sub> levels seem to be more tolerant to drought stress, potentially dampening this effect. Körner (2000) concluded that, besides a stimulation of photosynthesis, the most robust findings on plant responses to elevated CO<sub>2</sub> are changes in active tissue quality (wider C:N ratio) and effects on community dynamics. Kozovits et al. (2005) found that the type of competition (intra- versus inter-specific) changed the response of trees to elevated CO<sub>2</sub>. DeLucia et al. (2005) found an increase in NPP and NEP in both loblolly pine and deciduous sweetgum forests, but also found an increase in plant respiration that reduced NPP, more so in the pine than in the deciduous forest. DeLucia et al. (2005) warn that greater allocation to more labile tissues may cause more rapid cycling of carbon back to the atmosphere.

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The few studies that have modeled multi-factor influences on temperate forest net ecosystem productivity or carbon flux have found that combined effects are expected to diminish the effect of CO<sub>2</sub> enrichment alone.

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By far the vast majority of research has been done using single factor analyses. But biogeochemical processes and cycles, including carbon assimilation and flux, take place in a complex environment of changing climate, increasing atmospheric CO<sub>2</sub> and O<sub>3</sub>, nitrogen deposition, and varying land use legacies. The few studies that have modeled multi-factor influences on temperate forest net ecosystem productivity or carbon flux have found that combined effects are expected to diminish the effect of CO<sub>2</sub> enrichment alone. Scenario modeling of the combined influence of CO<sub>2</sub>, O<sub>3</sub>, temperature, and precipitation by Hanson et al. (2005) produced a 29% reduction in NEE over baseline conditions, even though models of CO<sub>2</sub> enrichment alone yielded substantial increases in NEE. Similarly, both Ollinger et al. (2002) and Zak et al., (2007) found that O<sub>3</sub> significantly dampened the fertilization effect of CO<sub>2</sub> and nitrogen. Models of historical forest growth and productivity as atmospheric CO<sub>2</sub> and nitrogen deposition increased in North America from 1700 to 2000 show that past agricultural land use depresses forest growth compared to past timber harvesting (Ollinger et al., 2002), which the authors attribute to depletion of soil nitrogen in agricultural lands.

More research is needed to elucidate changes in stand-level biogeochemical cycling with a focus on large-scale long-term experiments such as at FACE sites. As the literature shows, there is no clear answer as to whether rising CO<sub>2</sub> concentrations will cause forests to grow faster and store more carbon (Körner et al. 2005). The response to increasing atmospheric CO<sub>2</sub> will be confounded by the effects of prior land use and changes in temperature, precipitation, and radiation on forest productivity response.

## CONCLUSIONS

Currently the temperate forest biome is a small net sink for atmospheric carbon dioxide: however, that status rests on a tenuous balance between stable forest area, age-class distribution, disturbance regimes (windstorms, fire, insects, management), successional patterns, and the potentially counteracting effects of climate change and levels of atmospheric CO<sub>2</sub>, nitrogen and ozone. At best, if land use change remains in balance and forest productivity remains high, temperate forests will remain a small carbon sink. Significant changes in forest cover or age-class distribution across the biome, however, would shift temperate forests to being either carbon-neutral or a source of CO<sub>2</sub> emissions, further exacerbating climate change.

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## Chapter 6

# Carbon Dynamics of Boreal Forests

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### EXECUTIVE SUMMARY

As one of the largest and most intact biomes, the boreal forest occupies a prominent place in the global carbon budget. While it contains about 13% of global terrestrial biomass, its organic-rich soils hold 43% of the world's soil carbon. A growing body of research has attempted to measure how climate influences the processes governing carbon uptake and release, and to predict further changes due to climate change. A review of this body of research produces the key findings outlined below.

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Given what is presently understood about carbon pools in the boreal forest and the processes that affect them, it can be said that at present this forest biome acts as a weak sink for atmospheric carbon. However, the conditions that make this true are tenuous, and evidence of rapid climate change at northern latitudes has raised concern that the boreal forest could change to a net source if the ecophysiological processes facilitating carbon uptake are sufficiently disrupted. Changes in soil temperatures, respiration rates, and disturbance type, extent, and frequency brought about by climate change or other factors could switch the biome to a net source of carbon. Based on current knowledge, it appears that a warming climate will likely create the conditions for increased carbon release from boreal forests.

That being said, however, determining the balance of carbon uptake and release is highly complex, and methods of measurement will have to improve before any definite conclusions can be drawn about climate change impacts.

### What do we know about carbon storage and flux in boreal forests?

- Research indicates that boreal forests across North America and Eurasia have acted as weak sinks for atmospheric carbon in the last century. Storage of carbon in living and dead vegetation and the organic soil pool generally exceeded carbon release through respiration and combustion. The “sink”

status of the boreal forest is largely dependent on factors that keep heterotrophic respiration (release of CO<sub>2</sub> through decomposition of organic matter) lower than carbon uptake through plant growth and accumulation in the soil. Heterotrophic respiration varies with the amount of decaying organic matter, soil moisture, soil temperature, and vegetation type, which in turn are influenced by disturbance (particularly fire and insect outbreaks, but also harvesting and ice and wind storms), temperature, precipitation, and duration of thaw.

- The soil carbon pool plays a disproportionately large role in boreal forests, frequently constituting the largest pool in the system. In general, carbon storage rates in the soil are highest in low, saturated sites such as peat bogs or black spruce swamps. More productive, well-drained sites on uplands may produce greater tree growth but store less carbon in the soil pool.
- Studies in Canada have shown that lichens and bryophytes in lowland saturated sites contain upwards of 20% of the above ground carbon. These communities have important effects on how carbon is stored in boreal soils. Thick moss layers limit heat gain from the atmosphere, creating cold and wet conditions that promote the development of permafrost, with limited decomposition, thus are important for carbon storage.

#### **What don't we know about carbon storage and flux in boreal forests?**

- There is a tremendous amount of uncertainty in estimates of boreal carbon pools, because there have been so few studies compared to the vast extent of the biome, and most have been done in Canada and Fennoscandia.
- There is little quantifiable information about several important carbon pools, including fine root biomass and mycorrhizae; bryophyte and understory layers; and coarse woody debris and litter in Russia.
- Research is lacking on poorly drained sites, which may be the most vulnerable to soil carbon loss with changes in disturbance regimes and climate; and on Russian larch forests.
- Considering the importance of fire in boreal carbon dynamics, there is much that is not well understood, including extent, frequency, and intensity across the biome; and the interactions among fire intensity, nitrogen, and carbon.

#### **What are the major influences on carbon storage and flux in boreal forests?**

##### *Disturbance*

- Increased fire frequency could greatly increase carbon release, especially if it increases the decomposition of "old" carbon from the soil pool by increasing soil temperatures and degrading permafrost. More frequent fires could greatly reduce storage in woody biomass, and cause a concurrent increase in decomposition. Of even greater importance is the enhanced rate of

heterotrophic respiration observed after fire. This occurs because fire-killed trees begin to decompose after the event, and also because the fire removes the insulating bryophyte and litter layers that keep soil respiration low. In addition, fire regimes determine the forest age class distribution across the landscape, and influence what vegetation communities develop (with their differing carbon dynamics). On the other hand, an often-overlooked impact of fire is the conversion of woody biomass to charcoal, a very persistent form of carbon that can remain in the soil for centuries. Thus fire may actually enhance carbon storage in the soil by contributing to the charcoal pool.

- While fire is recognized as the dominant natural disturbance type over much of the boreal forest, insect outbreaks (and “background” insect damage during non-outbreak years) are also critically important. In some circumstances, such as the Canadian boreal and north temperate forests, insects and pathogens annually cause forest volume losses through mortality and growth reductions that are three times the volume lost to fire. Unlike fire, insect damage does not produce a direct emission, but rather exerts its influence through altered rates of decomposition and growth. In some forest types, insect outbreaks exert the primary influence on age class distribution.

#### *Age class distribution*

- The balance of carbon uptake versus respiration loss changes with the stage of stand development in boreal forests, and research indicates that two distinct scenarios may be possible. In the first more frequently observed scenario, a brief period of enhanced post-disturbance (fire or logging) release is followed by a return to sink conditions and, eventually, equilibrium. The “sink” status of boreal forests is thus dependent on a disturbance regime that creates a forest age-class distribution that is skewed towards vigorous, maturing stands. However, other research indicates that decomposition of post-fire detritus may not occur early in stand development, but rather during stand maturation. Such a delayed decomposition response could counteract the high carbon uptake rates observed in maturing stands, making them a weaker sink than traditionally thought.

#### *Temperature and precipitation*

- Extremely high rates of carbon storage are possible in many boreal soils due to insulating bryophyte layers, low temperatures, high moisture content and permafrost formation. The cold, wet conditions found in these soils slow decomposition rates and allow organic matter to accumulate faster than it is respired away.

#### **How might the carbon status of boreal forests change with changing climate?**

- The question of whether moisture availability will decline with climatic warming will probably determine whether warming enhances the boreal



carbon sink or turns it into a source. The balance of growth and respiration is significantly influenced by climatic conditions such as temperature, precipitation, and duration of the growing season. Increasing temperatures without concurrent increases in precipitation can cause drought stress, increased respiration, and the loss of carbon from boreal forests. However, if precipitation increases along with temperature, growth conditions could significantly improve and greater carbon uptake could occur. Increasing temperatures in early spring could also increase carbon uptake by lengthening the growing season.

- Sustained increased temperatures could possibly cause the breakdown of permafrost layers in boreal soils. If this occurs, the large stores of carbon bound in these frozen soils could be released.
- It appears that climatic warming is shortening the fire return interval in many boreal forests, and speeding up the life cycles of damaging insects. This could result in a large release of carbon, quickly turning the boreal forests from a sink to a source of carbon.
- Peatlands are possibly at greater risk from climate warming than forested areas and there is very little research on these unforested wetlands, which may hold the majority of the carbon found in the boreal system.
- Over 97% of the total carbon stored in the vast tundra systems to the north of the boreal forest is found in the soil. This has huge implications for the global carbon budget, with the potential for a shifting boreal-tundra border with climate change. It is unclear whether the massive soil pool in tundra sites would remain intact if converted to a forested biome.

## **INTRODUCTION**

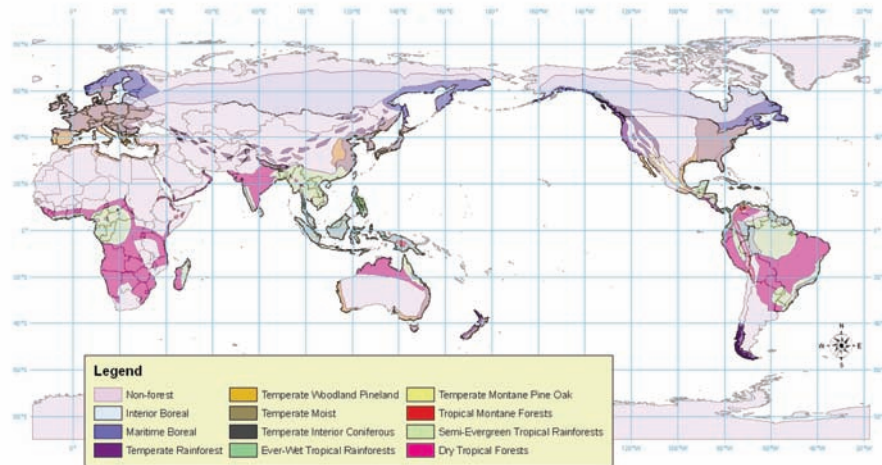
This chapter reviews the research literature on boreal and sub-boreal forests of Eurasia and North America. It first describes the region, the forest types, and their climatic variations. It then describes the stocks of carbon within the different components of the forest – above-ground biomass, below-ground biomass, lichens and bryophytes, the litter layer, and the soil. The next part of the chapter is focused on changes among carbon stocks – in particular understanding the biotic interactions of uptake (photosynthesis) and loss (respiration, decomposition); and then how abiotic influences of disturbance (fire, insect outbreaks, forest management) can affect carbon stocks. The chapter highlights areas of carbon forest science that we know versus those aspects that we do not know and those in which more work needs to be done.

### **The boreal forest system**

The boreal forest occupies a vast swath of the northern hemisphere, including much of Canada, Alaska, Fennoscandia, Russia, Mongolia, and northeast China (Figure 1).

Its northern limit is close to 68°N in North America and nearly 71°N in Eurasia, north of which tundra vegetation dominates. The southern limit is more variable, blending into temperate mixed forests or grassland and steppe systems, depending on moisture availability (Larsen, 1980). Certain temperate forests that border the boreal (such as the Laurentian forest types of eastern North America or the Ussuri Taiga of the Russian Far East) or that occur at high elevations (such as spruce-fir communities in the Rocky Mountains or the Alps) have similar dynamics of carbon storage and release, and much of the research cited in this paper can be applied to these regions.

**Figure 1** Original extent of boreal, temperate, and tropical forest types of the world prior to land clearing



Across their global range, boreal forests share certain key features. Only six tree genera are found as canopy dominants: spruce (*Picea*), fir (*Abies*), pine (*Pinus*), larch (*Larix*), birch (*Betula*), and aspen (*Populus*). Mature stands tend to exhibit very simple structure, dominated by a single stratum of conifers with a well-developed bryophyte layer at ground level (Gower et al., 2001). Understory communities tend to be depauperate (Larsen, 1980). In sub-boreal forests along the southern edge of the zone, aspens and birches may become more dominant, with a concomitant increase in understory diversity. Boreal landscapes in North America and Eurasia feature vast plains (often the beds of ancient glacial lakes) interspersed with numerous bogs and fens. These plains are bounded by mountain ranges such as the Northern Rockies and the Altai (Figure 1). Soils are predominantly heavily leached and nutrient-poor podzols (Larsen, 1980). In lowland areas with sufficient moisture and temperature conditions, large peat deposits form above the mineral soil, sometimes covering many millions of hectares (Gorham, 1991).

Differences in climate, moisture availability, and disturbance regimes create distinct zones within the greater boreal continuum. In North America, interior boreal forests occupy the majority of the area, characterized by a continental climate, dominance by white spruce (*Picea glauca*), jack pine (*Pinus banksiana*) and spruce-

aspen (*Populus tremuloides*) mixedwoods, and a disturbance regime of catastrophic fires. In contrast, maritime influence from the Pacific in the west, and the Atlantic Ocean in the east create moister, more productive conditions in the Cordillarean and Maritime boreal zones, respectively (Apps et al., 1993, Baldocchi et al., 2000) (Figure 1). These types include a greater component of fir species (*Abies* spp.) and are heavily influenced by industrial forest use and cyclical outbreaks of forest insects.

In Eurasia, boreal forests west of the Ural Mountains tend to be dominated by Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*), and are significantly influenced by catastrophic fire and industrial forest management practices. The Baltic and White Seas produce a moderating climatic effect for Fennoscandian and northwest Russian forests (Baldocchi et al., 2000) which may explain the higher productivity observed in these areas than in continental Siberian forests (Schulze et al., 1999) (Figure 1). East of the Urals, a combination of extreme moisture stress and extensive permafrost shifts the competitive advantage to larch species (*Larix* spp.), which are adapted to these difficult growing conditions (Gower and Richards, 1990). Large areas of Scots pine are also found in Siberia. A regime of frequent, non-catastrophic ground fires is characteristic of these forests (Harden et al., 1997).

## **POOLS OF CARBON IN THE BOREAL**

Carbon storage in the boreal forest occurs in distinct but interrelated pools, each of which demonstrates unique reactions to environmental stimuli. As such, it is very important to address these pools separately before attempting an integrated understanding of boreal carbon dynamics. The major pools are aboveground biomass (ranging from 11% to 59%); soil (ranging from 20% to 85%); and bryophytes/mosses (ranging from 5% to 26%) (Table 1). Litter and belowground biomass are much smaller, although the litter pool can be as high as 50% in young Jack pine stands. Belowground biomass is hard to measure and consequently there are limited data for this pool.

### **Aboveground biomass**

This pool consists of the live or dead standing biomass of trees, shrubs and herbs. In contrast with tropical and temperate forests, this aboveground pool is usually not the largest in the boreal system but is strongly influenced by site productivity. For example, in relatively productive upland aspen and jack pine sites in central Canada, aboveground vegetation and soil contained roughly equal amounts of carbon. In contrast, in lowland swamps of stunted black spruce (*Picea mariana*), only about 12-13% of the carbon was found aboveground (Gower et al., 1997). Black spruce stands in Manitoba had  $40 \pm 13$  tons carbon per ha (living and dead biomass), which comprised around 15-23% of total stand carbon depending on whether the sites were saturated swamps or well-drained uplands (Goulden et al. 1998). In south Siberia, biomass carbon exceeded soil carbon in Scots pine stands, while it was near equal in birch stands, and was exceeded by soil carbon in larch stands (Vedrova et al. 2002). In an

interior Canadian black spruce forest, Malhi et al. (1999) reported that aboveground biomass makes up on average around 11% of total stand carbon (see Table 1).

Overstory (tree) vegetation appears to dominate the aboveground pool of which approximately 5% may be dead trees (Yarie and Billings, 2002). The woody understory comprises a minor component of total forest carbon (Nalder and Wein, 1999; Li et al., 2003), and was measured in one study as less than 2% (Wang et al., 2001).

**Table 1** Distribution of carbon among different pools in boreal forests

Source	Site Characteristics			Carbon Pools			
	Location	Forest Type	Age	Aboveground Biomass	Bryophytes/Mosses	Litter	Soil
Malhi et al., 1999	Interior Canada	Black Spruce	115	49.2 Mg/ha (11%)		6.2 Mg/ha (1%)	390.4 Mg/ha (85%)
Goulden et al., 1998	Interior Canada	Black Spruce Sphagnum site	120	40 ± 20 tons /ha (14%)	45 ± 13 tons /ha (16%)		200 ± 50 tons /ha (70%)
Goulden et al., 1998	Interior Canada	Black Spruce Pleurozium site	120	40 ± 20 tons /ha (23%)	45 ± 13 tons /ha (26%)		90 ± 51 tons /ha (70%)
Gower et al. 1997	Interior Canada	Black spruce	115 - 155	49.2 - 57.2 Mg /ha (11 - 12%)			390.4 - 418.4 Mg /ha (87 - 88%)
		Aspen	53 - 67	57.0 - 93.3 Mg /ha (32 - 59%)		15.9 - 19.4 Mg/ha (9 - 12%)	36.0 - 97.2 Mg/ha (23 - 55%)
		Jack pine	25	7.8 - 12.3 Mg/ha (10 - 24%)		18.1 - 40.3 Mg/ha (36 - 53%)	20.2 - 28.4 Mg/ha (37 - 40%)
		Jack pine	65	29.0 - 34.6 Mg/ha (42 - 51%)	3.5 - 5.1 Mg/ha (5 - 7%)	11.5 - 14.6 Mg/ha (17 - 21%)	14.2 - 25.8 Mg/ha (20 - 38%)

Aboveground productivity in boreal forests is limited by a number of environmental factors, including seasonal distribution of precipitation, timing of soil thaw, soil type, nutrient availability, site aspect, topography, and length of the growing season (Gower et al., 2001). Many of these factors affect productivity primarily by controlling rates of respiration and decomposition, which will be explained further in the sections on “Drivers of Uptake and Release.” One example, nitrogen availability, is often identified as a growth limitation in boreal forests (Bonan and Van Cleve, 1992). This limitation may be linked to very slow decomposition rates which trap nitrogen in undecomposed litter (Wirth et al., 2002). Thus, decomposition and its drivers (soil warming, water table depth, forest fire) determine the extent to which nitrogen limits aboveground productivity.

Aboveground carbon storage also appears to differ across forest types. It is greater in mixed woods than pure stands of either deciduous or coniferous trees, perhaps due to the greater foliage mass in stratified mixed stands (Martin et al., 2005). Aboveground and total net primary production (NPP) are generally higher in deciduous

than coniferous stands (Gower et al., 1997; Gower et al., 2001), but this will not necessarily lead to higher rates of carbon uptake if accompanying respiration is also higher.

Research from the Russian taiga indicates that disturbance and extreme climatic events (i.e. drought) may prevent boreal forests from attaining the maximum density and productivity possible under site conditions (Schulze et al., 1999; Vygodskaya et al., 2002). For instance, south Siberian forests were kept below the theoretical self-thinning line by frequent ground fires that reduced stand density beyond the levels associated with competition mortality (Schulze et al. 1999). The importance of such events must be considered along with site factors in quantifying the aboveground carbon pool.

### **Belowground biomass**

The belowground biomass carbon pool consists of coarse and fine tree roots and their associated mycorrhizae. It is considered one of the most difficult pools to quantify, as labor-intensive destructive sampling is often required to achieve exact figures, and even then measuring fine root mass may not be possible (Table 2). Gower et al. (2001) found that the most common bias in estimations of NPP in boreal forests was the exclusion of fine roots and mycorrhizae from the calculation. The few studies that have measured these features show high variability and thus cannot be extrapolated to quantify the belowground pool for the biome.

While precise quantification of belowground biomass is difficult, researchers have been able to identify the approximate proportion of total stand carbon that this pool accounts for (Table 1). Data from limited studies show that belowground biomass is highly variable, influenced by such stand and site factors as species composition, stand age, and available moisture. A greater percentage of total NPP is allocated to roots in coniferous than in hardwood stands (Bond-Lamberty et al., 2004). One study found that 41- 46% of total NPP was allocated to roots in conifer stands but only 10-19% in aspen stands (Gower et al., 1997). However, research in Alaska has shown that hardwood forests can exceed coniferous forests in the production of fine roots, which can make up 11-29% of stand biomass (Ruess et al., 1996). Stand age appears to affect the belowground biomass pool by regulating root production. Bond-Lamberty et al. (2004) found that coarse and fine root production peaked at around 70 years in a Canadian black spruce chronosequence, but was 50-70% lower in 151 year old stands.

Soil moisture limitations may cause trees to allocate more biomass to belowground structures. Schulze et al. (1999) found that a greater proportion of stand biomass was allocated to roots in Siberian boreal forests than in European Russia or temperate European forests, perhaps due to the extreme moisture deficits that occur in some areas of Siberia. Indeed, increasing aridity across northern Siberia may be causing a shift in allocation from photosynthetic tissues to roots, while increasing moisture in European Russia and south Siberia is having the opposite effect (Lapenis et al., 2005). Other environmental factors besides moisture could also be at play here: Prokushkin et al. (2005) attributed the high relative allocation to roots in Siberian forests to low soil temperatures and nutrient availability. It appears that under stressful conditions with low levels of water and nutrients, trees develop larger root systems to access these resources.

**Table 2 Sources of uncertainty in boreal carbon modeling.** The following summarizes portions of the boreal carbon budget (pools, processes and environmental variables) that are currently poorly understood or quantified. This indicates directions for future research on boreal carbon dynamics.

Inadequately quantified carbon pools	Reference
Fine root biomass/mycorrhizae	Gower et al., 1997, 2001
Magnitude of labile soil carbon pool	Rustad and Fernandez, 1998; Jarvis and Linder, 2000; Bronson et al., 2008
Bryophyte/understory layers	Gower et al., 2001
CWD and litter in Russia	Krankina et al., 2002
Changing allocation patterns within trees	Lapenis et al., 2005

Poorly understood environmental variables	Reference
Quantifying burned area in Russia	Dixon and Krankina, 1993; Conard and Ivanovna, 1997; Soja et al., 2007
Recognizing refugia in burned areas	Amiro et al., 2001; Kang et al., 2006
Fire intensity vs. simply fire occurrence	Wooster and Zhang, 2004
Influence of burn severity on carbon and nitrogen consumption	Balshi et al., 2007
Accounting for ground vs. crown fires	Wirth et al., 2002
Changes in insect life cycles	Malstrom and Raffa, 2000
Possibility of poor post-disturbance stocking	Auclair and Carter, 1993; Shvidenko et al., 1997
Accounting for potential vegetation dieback	Kasischke et al., 1995
Rates of permafrost degradation	Prokushkin et al., 2004
Lag time on migration of temperate species into boreal zone	Smith and Shugart, 1993
Quantifying area, depth and bulk density of boreal peatlands	Gorham, 1991
Balance of CO <sub>2</sub> and CH <sub>4</sub> emissions from peatlands	Gorham, 1991
Lack of research on poorly-drained forests	Bond-Lamberty et al., 2004
Rates of precipitation change	Pastor and Post, 1988; Flannigan et al., 1998
Accuracy of estimation of crown and soil temperatures	Arain et al., 2002
Varying temperatures of different carbon pools	Lindroth et al., 1998
Assumption of increased productivity with increased temperature	Briffa et al., 1998; Barber et al., 2000; Wilmking et al., 2004
Timing of increased temperatures	Lindroth et al., 1998
Using monthly temperature anomalies as opposed to daily temperature data	Flannigan et al., 1998
Thresholds in NEP response to climate change	Grant et al., 2006
Albedo effect of boreal forest cover	Bonan et al., 1992, 1995; Betts, 2000; Bala et al., 2007
Lack of data on Eurasian larch forests	Gower et al., 2001

### Lichens and bryophytes

This pool is composed of lichens, bryophytes, and mosses, which frequently form a dense mat at the ground level in boreal forests. This pool is uniquely important to

boreal forests while, in contrast, it is an insignificant component of the carbon budget in temperate and tropical zones.

Soil drainage seems to influence the magnitude of this pool (Turetsky et al., 2005), which is largest in boreal peatlands, where bryophytes are the major vegetation type. In mature lowland black spruce forests, mosses may sequester as much or more carbon than trees, and 10 times the amount sequestered by understory vegetation (Harden et al., 1997). Czimczik et al. (2006) found that bryophytes made up 20% of total aboveground NPP in black spruce stands. The dominant bryophytes in such saturated sites are *Sphagnum* mosses. In upland spruce sites with better drainage, the moss dominance switches to *Pleurozium* feathermosses, which accumulate significantly less carbon than *Sphagnum* types (Goulden et al., 1998). Moving even further “upland,” only 3.2% of stand carbon is stored in mosses in xeric jack pine stands, and in aspen stands the bryophyte pool is even smaller (Nalder and Wein, 1999).

Unfortunately, no research on the importance of bryophytes in Eurasian boreal forests was found for this review. Given the circumpolar range of *Sphagnum* and *Pleurozium* species, and the widespread presence of saturated lowland boreal forests in Eurasia, it seems likely that bryophytes also play a large role in that region. Little is also known about the dry lichen communities (often composed of *Cladonia* species) that blanket the floor of xeric conifer woodlands in North America and Eurasia. Despite recognition of their unique importance, lichens and bryophytes remain one of the least studied carbon pools in the boreal forest (Table 2).

In addition to their direct role as a carbon pool, bryophyte communities have important effects on how carbon is stored in boreal soils. Thick moss layers (including live mosses and moss-derived organic material) in Canadian black spruce stands limit heat gain from the atmosphere, creating cold and wet conditions near the soil surface that promote the development of permafrost. The limitations on decomposition imposed by such conditions are very important for carbon storage in the soil profile. In white spruce and aspen stands with less-developed bryophyte communities, more rapid transfer of heat, moisture, and oxygen through the soil profile is possible, resulting in warmer and drier subsoil conditions and less stored carbon (O'Neill et al. 2002).

The flammability of different bryophyte communities influences their rates of carbon storage and release. *Pleurozium* mosses dry out completely; consequently, a fire can release the carbon stored therein and expose the soil surface to greater heat and drying. In contrast, *Sphagnum* mosses remain saturated through most of their profile, even during dry seasons. Fires only remove the upper layers, leaving moist lower layers intact to insulate the soil (Harden et al. 1997). The reduced flammability and decomposition brought about by *Sphagnum* communities contribute to the general trend of greater ground-level and belowground carbon storage in saturated lowland sites than in well-drained uplands.

### Litter layer

The litter pool is made up of dead organic matter that is not completely decomposed and has not yet entered the soil profile. Malhi et al. (1999) found that the litter layer

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Despite recognition of their unique importance, lichens and bryophytes remain one of the least studied carbon pools in the boreal forest.

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composes on average only about 1% of total stand carbon in boreal forests (Table 1). The size of this pool is primarily driven by rates of decomposition and disturbance. Disturbances such as fire or insect infestation contribute pulses of dead material to the pool, but fire can also reduce it through direct burning or by raising ground temperatures and stimulating increased decomposition. Young post-fire stands often have very large litter pools (composed of the dead remains of the previous cohort), which eventually decompose, depleting the pool in maturing stands. Increased overstory mortality with increasing age can gradually replenish the supply of litter. This sequence of depletion and re-accumulation demonstrates that there is no simple relationship between litter carbon and stand age. In fact, the forest floor of Canadian jack pine stands was shown to lose carbon with age (Nalder and Wein, 1999).

Rates of litter accumulation vary across boreal zones. In Russian boreal forests, the differences may be caused by composition. Stocks of coarse woody material are greater in Siberia, where rot-resistant larch species predominate, than in pine- and spruce-dominated European Russia (Krankina et al., 2002). Nalder and Wein (1999) found that the density of forest floor carbon was 68% higher in jack pine stands in eastern Canada than in western Canada. The reasons for such differences across the same vegetation community are not entirely clear. Differing site productivity, decomposition rates or fire levels could be involved.

The litter layer also interacts with bryophyte communities to affect soil properties. Like mosses, thick litter layers can insulate the soil, affecting depth of thaw, available moisture and belowground respiration (Bonan et al., 1990). The insulating and moisture-retaining capacity of the forest floor (including both litter and bryophytes) is highest in black spruce forests among all Canadian boreal forest types (Van Cleve et al., 1990). In such stands, the combined litter-bryophyte “ground” layer may store three to four times the carbon held in aboveground biomass (Kasischke et al., 1995).

### Soil carbon

The soil pool (found below the litter layer, consisting of decomposed organic matter and mineral soil) is the most important in the boreal carbon budget. The amount of carbon held in the soil profile often dwarfs that in the forest vegetation (Malhi et al., 1999; Goulden et al., 1998; Kasischke et al., 1995; Wirth et al., 2002), a unique feature of the boreal forest (Table 1). Many of the same factors responsible for carbon accumulation in bryophyte and litter layers help explain the prominence of soil carbon: cold, saturated soils have low rates of decomposition, allowing carbon-rich organic matter to accumulate in the soil profile faster than respiration losses. Thus, the soil pool is largest in the most saturated sites. Unforested wetlands may hold the majority of the carbon found in the boreal system, significantly out of proportion to their position in the landscape (Kasischke et al., 1995; Rapalee et al., 1998). For example, lowland (*Sphagnum* site) black spruce soils contain  $200 \pm 50$  tons carbon per ha, while upland (*Pleurozium* site) soils contain only  $90 \pm 20$  tons per ha (Goulden et al., 1998). Soil carbon storage in well-drained (and more productive) aspen and jack pine stands is 2.8-2.9 times smaller than in saturated black spruce soils, which contain 87-88% of stand carbon (Gower et al., 1997). In xeric Scots pine



stands in Siberia, by contrast, biomass carbon may exceed soil carbon (Vedrova et al., 2002; Wirth et al., 2002).

The dominant position of belowground carbon is even more pronounced in the tundra systems to the north. Over 97% of the total carbon stored in these systems is found in the soil (Billings, 1987). This has huge implications for the global carbon budget with the potential for a shifting boreal-tundra border with climate change. It is unclear whether the massive soil pool in tundra sites would remain intact if converted to a forested biome (Kasischke et al., 1995).

Where carbon is found in the soil profile also varies. In saturated black spruce sites, it is often found in the organic horizons or directly below (Goulden et al., 1998; O'Neill 2002), while the majority of carbon in upland aspen (92%) and white spruce (82%) soils are found in mineral soil (O'Neill, 2002). However, in upland larch (*Larix gmelinii*) forests in northeast China, soil carbon concentration decreases significantly with soil depth across a range of mesic to xeric sites. This may be attributable to pulses of charcoal added to upper layers by recent fires (Wang et al., 2001).

Fires appear to be very important for transferring carbon from vegetation to the soil profile through conversion to charcoal, a decay-resistant form that can reside in the soil 3,000-12,000 years (Deluca and Aplet, 2008). While some is worked down into lower soil horizons by cryoturbation (mixing of soil layers by the freeze-thaw process) (Hobbie et al., 2000), the large majority remains above 30 cm in depth, with approximately 70% remaining above 10 centimeters (Deluca and Aplet, 2008). One study estimated that 30% of the biomass killed in a fire enters the soil as charcoal or unburned material, at least half of which may enter the long-term soil pool; the rest is lost to decomposition or re-burning over the next century (Harden et al., 1997). In the Rocky Mountains, charcoal can make up as much as 60% of soil carbon (Deluca and Aplet, 2008), while in south Siberia this figure is 20-24% (Schulze et al., 1999).

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Unforested wetlands may hold the majority of the carbon found in the boreal system, significantly out of proportion to their position in the landscape.

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## BIOTIC DRIVERS OF UPTAKE AND RELEASE

Biosphere-atmosphere carbon flux consists primarily of three processes: photosynthesis, autotrophic respiration, and heterotrophic respiration (through decomposition of organic matter). Along with biomass burning, these processes determine the balance between uptake and release of carbon from forests.

### Photosynthesis and autotrophic respiration

These two processes are paired because they essentially represent opposite forces acting on the carbon budget. Carbon uptake by photosynthesis must be paired with carbon loss through autotrophic respiration, which consumes 54-77% of annual net photosynthesis in boreal forests (Ryan et al., 1997). While autotrophic and heterotrophic respiration are often considered together (due to the difficulty of distinguishing them during measurement), only the former is closely paired with photosynthesis. Heterotrophic respiration rates are not necessarily proportional to tree growth (Li et al., 2003; Barr et al., 2007).

The pairing of photosynthesis and autotrophic respiration does not imply that they necessarily respond the same way to environmental stimuli. In one study in a mature Canadian aspen forest, interannual variability of photosynthesis was controlled primarily by growing season length and secondarily by drought, whereas interannual variability in respiration was primarily controlled by drought and secondarily by temperature (Barr et al., 2007). Jarvis and Linder (2000) support the idea that canopy duration (i.e. length of growing season as controlled by spring temperature) is more important in determining total photosynthesis levels than average temperature or soil moisture levels. Indeed, 20<sup>th</sup> century increases in spring temperatures attributed to rising atmospheric CO<sub>2</sub> levels may have increased productivity in boreal aspen stands by allowing for earlier leaf out (Chen et al., 1999).

Rising temperatures (especially if encountered in early spring) may stimulate increased photosynthesis, but they also cause a rise in autotrophic respiration. Respiration rates rise faster under rising temperatures than photosynthesis rates, potentially causing carbon release to the atmosphere (Lindroth et al., 1998). Many models of boreal carbon flux assume that respiration responds directly to rising temperature, while photosynthesis is limited by other factors such as light levels, length of growing season, and water and nutrient availability. However, in a study of these processes in Canadian peatlands, increasing annual temperature was unexpectedly correlated with increased net carbon uptake, suggesting that photosynthesis may be more responsive than previously thought, and that respiration will not necessarily overtake it in a warming climate (Dunn et al., 2007).

The unexpected results of the above study may have been related to the abundant soil moisture available in peatlands. In drier upland forests, soil moisture availability imposes limitations on forest productivity (Chen et al., 1999; Gower et al., 2001; Bond-Lamberty et al., 2007). Rising temperatures unaccompanied by increasing precipitation could cause moisture stress, reducing photosynthesis. But importantly, drought also lowers respiration levels, potentially balancing out the reduced carbon uptake (Barr et al., 2007). The duration and severity of drought is important because mild drought suppresses respiration but leaves photosynthesis largely unchanged, while severe drought suppresses both, with a dramatic drop in photosynthesis levels as it intensifies (Barr et al., 2007).

### **Heterotrophic respiration and decomposition**

Heterotrophic respiration, caused by decomposition of organic matter in the soil and litter layers, is the largest source of carbon emissions in the boreal system. Conceptually, decomposition and organic matter accumulation act as opposite influences on the soil and litter carbon pools; if decomposition exceeds organic inputs, there is a net loss of carbon from the system (Harden et al., 1997). Heterotrophic respiration is a large enough component of carbon flux that it might offset not only organic matter accumulation, but also carbon gains from photosynthesis. Indeed, because photosynthesis and autotrophic respiration often

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Fires appear to be very important for transferring carbon from vegetation to the soil profile through conversion to charcoal, a decay-resistant form that can reside in the soil 3,000-12,000 years

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rise and fall together, the real determinant of whether a stand is a carbon sink or source may be its rate of heterotrophic respiration.

Certain environmental factors determine this rate. Vegetation type influences respiration rates through the differing qualities of litter produced. For instance, softwood litter decomposes slower than hardwood litter due to its high lignin content (Hobbie et al., 2000), and larch coarse woody material contains chemicals that slow the rate of rot relative to other softwoods (Krankina et al., 2002). Soil moisture exerts an even stronger influence on soil respiration rates (Harden et al., 2000). The high heat capacity of water and thick mats of bryophytes slow the warming of saturated soils. These factors limit baseline respiration rates, and also tamp down large spikes in respiration that follow fires (Harden et al., 1997). This explains the overall trend of higher soil carbon storage in lowland boreal forests than in upland forests. However, the constant saturation that limits release of CO<sub>2</sub> in boreal peatlands also promotes the release of methane (CH<sub>4</sub>), an important greenhouse gas. Drying of peatlands would have the opposite result, namely, decreased CH<sub>4</sub>, but increased CO<sub>2</sub> emissions (Gorham, 1991). This dynamic could become an important element of carbon flux under changing climatic conditions.

Soil temperature may be even more limiting to decomposition rates than soil moisture (O'Neill et al., 2002). Temperature is especially important in determining rates of winter respiration, a frequently overlooked process that may make up 20% of yearly respiration (Hobbie et al., 2000). Young deciduous stands that are carbon sinks during the growing season may become sources after senescence due to winter respiration (Pypker and Fredeen 2002, Trofymow et al. 2002). Such respiration appears to take place in deeper soil layers where temperatures remain high enough in the winter to support decomposition (Goulden et al., 1998). The organic matter in these layers is generally much older, less mobile carbon than that which is decomposed in the summertime (Winston et al., 1997; Dioumaeva et al., 2002). The temperature and duration of thaw in these soil layers control the decomposition rate of "old" soil carbon. Whether sustained soil warming associated with climate change would cause significant increases in carbon flux from this long-term pool is unclear.

Many studies have attempted to quantify how the balance of decomposition and vegetative growth shifts across a post-disturbance chronosequence (Figure 2). Increased respiration after a fire can be a significant source of carbon release. In fact, research has shown that post-fire decomposition may equal (Amiro et al., 2001) or exceed (Auclair and Carter, 1993) direct emissions from burning. Fire has a short-term impact on heterotrophic respiration rates by raising soil temperatures, stimulating increased decomposition of soil organic matter (Harden et al., 1997). There is a longer-term respiration response as well, when the trees killed by the fire begin to decompose a few years later. This process can potentially make young post-fire stands a source of carbon despite the vigorous regrowth of trees and mosses (Rapalee et al., 1998; Vedrova et al., 2002; Wirth et al., 2002). Similarly, in a chronosequence of post-harvest stands in central Canada, Li et al. (2003) found that stands younger than 20 years were carbon sources (releasing 193–239 g carbon/m<sup>2</sup> per year), but by 40 years of age had become weak sinks as growth outpaced

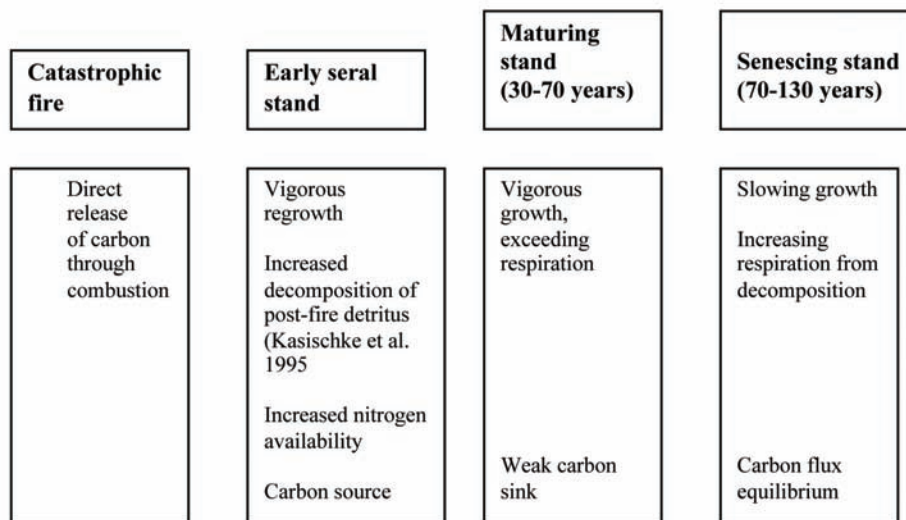
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In Canadian peatlands, increasing annual temperature was unexpectedly correlated with increased net carbon uptake, suggesting that photosynthesis may be more responsive than previously thought, and that respiration will not necessarily overtake it in a warming climate.

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decomposition. However, a post-fire chronosequence from the same region showed that significant decomposition of fire-killed litter did not occur in the first few decades and that young stands showed the lowest levels of respiration (Litvak et al., 2003). Czimczik et al. (2006) also did not observe a rise in decomposition in young post-fire stands in Canada. In fact, heterotrophic respiration did not become significant until black spruce dominated the canopy (around 70 years post-fire). These examples demonstrate that disturbance effects on decomposition rates may lag and occur later in stand development, and depend on the type of disturbance. It is also worth noting that increased heterotrophic respiration in young post-disturbance stands would be somewhat balanced by a decrease in autotrophic respiration, caused by tree mortality (Wang et al., 2001).

**Figure 2 Model of carbon dynamics through stand development in a Canadian black spruce forest.**



The constant saturation that limits release of CO<sub>2</sub> in boreal peatlands also promotes the release of methane (CH<sub>4</sub>), an important greenhouse gas. Drying of peatlands would have the opposite result, namely, decreased CH<sub>4</sub>, but increased CO<sub>2</sub> emissions. This dynamic could become an important element of carbon flux under changing climatic conditions.

Source: Derived from Litvak et al. 2003 unless otherwise noted in the figure.

### **DISTURBANCE: ABIOTIC DRIVERS OF UPTAKE AND RELEASE**

Disturbances such as fire, insect and pathogen outbreaks, and logging have important impacts on the boreal carbon budget. Disturbances influence the size of carbon pools by directly destroying (fire) or removing biomass (logging) from the system, and by altering the rates of photosynthesis and respiration as discussed earlier. In fact, disturbance may be the overriding factor in whether or not the boreal forest is a source or sink of carbon. For instance, Kurz et al. (2008) have estimated that large-scale insect outbreaks have turned Canada's managed forests from a carbon sink to a carbon source. Using Monte Carlo simulations, they predict that trend will continue due the effects of natural disturbances.

## Fire

The direct emission of carbon to the atmosphere through combustion is a significant component of boreal carbon flux. In upland sites in boreal Canada, Harden et al.'s (2000) model of long-term carbon balance estimated that 10-30% of the annual carbon production has been released as fire emissions, while 40-80% has been released during decomposition and 8-30% fixed as soil carbon. This estimate fits with other observations that increased post-fire decomposition has a greater impact than direct fire emissions (Auclair and Carter, 1993, Conard and Ivanovna, 1997). Quantifying direct emissions is a complicated task, beginning with the process of identifying the area burned in a given year across the vast boreal landscape. Underestimation of burnt area in Russia can significantly bias models, potentially missing a vital source of emissions to the atmosphere (Dixon and Krankina, 1993). In contrast, satellite estimation of forest fire extent in Canada overestimated cumulative burned area by approximately 22% because unburned inclusions were not recognized (Kang et al., 2006). These examples demonstrate the difficulty of accurately calculating this component of carbon flux.

As discussed earlier, fire affects soil properties through changes in temperature and moisture conditions, removal of insulating litter and bryophyte layers, and contribution of decay-resistant charcoal to the soil pool. Fire may also increase nitrogen input from the organic layer to the soil, increasing nitrogen mineralization and vegetation productivity (Kasischke et al., 1995; Johnson and Curtis, 2001; Kang et al., 2006). One study in the Canadian boreal demonstrated that deciduous stands are able to respond more rapidly to the increased supply of nitrogen than conifers, due to their faster rate of leaf canopy turnover. Thus, deciduous forests exhibited increased productivity with increasing fire frequency, while the opposite was true of both dry and wet coniferous types (Kang et al., 2006).

Across much of the boreal region, fire exerts a dominant influence on forest age class distribution. Fire-prone landscapes are characterized by a mosaic of age classes, each with differing rates of growth and respiration. Boreal carbon budgets must account for the different patterns of carbon uptake and release that accompany different age class distributions. In Canadian black spruce forests, most of the net biomass accumulation appears to take place from 20 to 70 years after a fire. Stands younger than 20 years lack sufficient leaf area for rapid carbon accumulation and stands older than 70 years are at or near zero carbon balance with the atmosphere (Figure 2). Only a small proportion (9%) of the black spruce stands in central Canada are in the most productive age class (around 36 years old) (Litvak et al., 2003). In boreal Quebec, biomass increased from 27-75 years following a fire, and decreased thereafter due to stand degradation. In the Alberta Boreal Plains ecoregion, it took between 15 to 30 years for post-fire stands to attain the same photosynthetic rates as mature areas while biomass continued to increase to at least 60 years of age (Amiro et al., 2000). Kasischke et al. (1995) reported, however, that biomass levels in upland black spruce forests in Alaska and northwest Canada continue to increase for 140-200 years after a fire, before increased overstory mortality sets in.

Such growth rate comparisons across stand age must be paired with rates of post-

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In fact, disturbance may be the overriding factor in whether or not the boreal forest is a source or sink of carbon. For instance, Kurz et al. (2008) have estimated that large-scale insect outbreaks have turned Canada's managed forests from a carbon sink to a carbon source. Using Monte Carlo simulations, they predict that trend will continue due the effects of natural disturbances.

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fire decomposition. In Siberian Scots pine forests, young post-fire stands are sources of carbon, and may take 70 years to reach pre-fire carbon levels (Wirth et al., 2002). Canadian studies also point to high initial rates of decomposition (Li et al., 2003; Litvak et al., 2003), although this trend may not always hold. Using eddy covariance measurements of growing season net ecosystem CO<sub>2</sub> exchange, Litvak et al. (2003) estimated that recently disturbed black spruce stands in Canada are sources of carbon, middle-aged (20-70 years old) stands are sinks, and older (70-130 years old) stands in near balance with the atmosphere. In Siberia, the trajectory is somewhat different: an initial decrease in carbon pools during first 30-40 years after a fire, fairly rapid carbon accumulation over the next 50 years, and lower but steady rates of accumulation in the centuries thereafter (Wirth et al., 2002).

The frequency and intensity of fire determines how forest age classes are distributed in many boreal landscapes (Table 3). In boreal forests of North America, Fennoscandia and European Russia, fires tend to be high-intensity and stand-replacing (Harden et al., 2000), and have a return interval of 40-110 years (Amiro et al., 2000). In Siberia, ground fires that are not stand-replacing are the norm, accounting for about 80% of the area burned. Such fires may burn through Scots pine stands on a short 25-50 year return interval, and larch stands on a 90-130 year interval, leaving many live trees. However, intervals seem to be considerably longer for spruce/fir stands, with fires in this type more likely to be catastrophic (Conard and Ivanovna, 1997). The total number of fires and the area burned are higher in Siberia than in North America, but the lower intensity of these fires means that more carbon is not necessarily released (Wooster and Zhang 2004). Models that fail to consider that detail can overestimate carbon emissions from Russian forest fires.

**Table 3** Fire regimes in the boreal forest

Forest Type/Location	Disturbance Type	Return Interval	Reference
<i>Pinus sylvestris</i> , NW Russia	Ground fire	20-40 years	Gromtsev, 2002
<i>Pinus sylvestris</i> , Siberia	Ground fire	25-50 years	Conard and Ivanovna, 1997
<i>Larix sibirica</i> , Siberia	Ground fire	90-130 years	Conard and Ivanovna, 1997
<i>Picea abies</i> , NW Russia	Stand-replacing fire	130-200 years	Gromtsev, 2002
Dark taiga <sup>1</sup> , central Siberia	Stand-replacing fire	400-500 years	Schulze et al., 2005
Continental taiga <sup>2</sup> , interior Canada	Stand-replacing fire	40-110 years	Amiro et al., 2000
Spruce/fir/birch <sup>3</sup> , eastern Canada	Stand-replacing fire	136 + 29 years	Lesieur et al., 2002
Boreal/tundra interface <sup>4</sup> , NW Canada	Stand-replacing fire	110 years	Johnson and Rowe, 1975

<sup>1</sup>*Picea obovata*, *Abies sibirica*, *Pinus sibirica*

<sup>2</sup>*Picea glauca*, *P. mariana*, *Pinus banksiana*, *P. contorta*, *Populus tremuloides*

<sup>3</sup>*Picea glauca*, *P. mariana*, *Abies balsamea*, *Betula papyrifera*

<sup>4</sup>*Picea glauca*, *Pinus banksiana*, muskeg vegetation

Rare, stand-replacing fires have different impacts on carbon dynamics than frequent ground fires. The post-fire chronosequences described above tend to occur in catastrophic fire systems, in which the aftermath of fire is nearly always mass mortality and decomposition, and a return to early-successional condition. Ground fires have a more complex result. They can produce uneven-aged communities

(Harden et al., 2000), and cause multiple small pulses of mortality and decomposition within the same stand. Rather than causing sudden, complete changes in stand development, ground fires alter competition and productivity levels within the existing cohort. Low-intensity fires in Siberian Scots pine stands result in a 10-20 year growth depression of the surviving trees due to fire damage, followed by 10-15 years of accelerated growth under reduced competition and higher nutrient supply (Schulze et al., 1999). In this forest type, young growth does not appear to necessarily replace the trees lost to ground fires. Instead, low-density stands persist and may never attain the maximum possible stocking (Schulze et al., 1999; Wirth et al., 2002). This “lost” productivity has a significant impact on carbon uptake in Siberian forests; Shvidenko et al. (1997) calculated a 45-50% reduction in forest productivity due to ground fires across large areas of Siberia.

Suppression of forest fires also affects the carbon budget. For example, temperate oak (*Quercus*) forests under fire suppression management had 90% more total ecosystem carbon than those with a frequent fire regime (Tilman et al., 2000). If fire suppression is practiced across a significant portion of the landscape, pools of biomass and litter carbon may exceed estimates for forests under a natural fire regime (Price et al., 1997). However, there is an inherent danger in fire suppression because larger fuel loads may, if ignited, produce much more intense fires than might have occurred in a natural fire regime.

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### **Insect outbreaks**

While fire is recognized as the dominant natural disturbance type over much of the boreal forest, insect outbreaks (and “background” insect damage during non-outbreak years) are also critically important. Across the Canadian boreal and north temperate forests, insects and pathogens annually cause forest volume losses through mortality and growth reductions that are three times the volume lost to fire. Malstrom and Raffa (2000) found that insects are especially dominant in the moist eastern regions of Canada. Indeed, in the balsam fir (*Abies balsamea*) dominated forests of the Maritime Provinces, cyclical outbreaks of the defoliating insect spruce budworm (*Choristoneura fumiferana*) supplant fire as the primary influence on age class distribution (Baskerville, 1975). Unlike fire, insect damage does not produce a direct emission, but rather exerts its influence through altered rates of decomposition and growth (Kurz et al., 2008).

Kurz et al. (2008) modeled the impact of spruce budworm and western mountain pine beetle (*Dendroctonus ponderosae*) outbreaks on carbon flux in the Canadian forest. They concluded that these events could switch the region from a carbon sink to a source due to the massive increases in decomposition of dead trees that follow outbreaks. Background levels of insect herbivory are also important. In Fennoscandian and Russian birch (*Betula pubescens*) forests, defoliating insects had a significant effect on leaf area index and net primary production. If certain levels of herbivory are reached, coniferous species may take over the growing space relinquished by damaged birches, accelerating stand development and causing related changes in carbon dynamics (Wolf et al., 2008). The combination of drought and

defoliating insects can result in significantly reduced production in Canadian aspen forests. If climate change results in an increase in drought and insect outbreaks, closed aspen forests may transition to sparse parklands (Hogg et al., 2002).

### Forest management

Besides its impacts on growth and decomposition rates, the commercial harvest of trees has a direct impact on carbon stocks through the removal of biomass from the forest. The eventual decomposition or combustion of this pool must be considered (refer to Chapters 12 and 13 for an analysis of wood products). The greatest difference between timber harvesting and other disturbance types is in the altered contribution it makes to the litter pool compared to fire or insect outbreak. Logging adds litter in pulses that are concentrated around harvest events, and the litter tends to lack stemwood, which is removed from the site for forest products. Intensive site preparation techniques, such as slash burning, can limit this pool even further. Krankina et al. (2002) found that intensively managed European Russian forests had much larger stocks of coarse woody material than unmanaged Siberian forests of similar productivity.

Field studies by Martin et al. (2005) suggest that the stand-level impacts of logging on soil carbon dynamics are limited. Harvesting has no consistent effect on carbon levels in soil detritus. Johnson and Curtis (2001) came to a similar conclusion, although they found that whole-tree harvests (as opposed to stem-only harvests that leave tree crowns in the forest) could cause slight decreases in soil carbon. However, long-term modeling of managed boreal forests shows a consistent decline in soil carbon across a 300-year time period compared to forests under a natural disturbance regime (Seely et al., 2002). Long term research plots in managed forests will be necessary to determine if this prediction is accurate.

Timber harvesting is concentrated in certain regions of the boreal forest. Fennoscandia and Maritime Canada are under near-complete management, while vast swathes of interior Canada and Siberia have experienced virtually no logging (although this could change in coming decades). Thus the impacts of forest management on the boreal carbon budget are uneven and difficult to compare with natural disturbances. In south Siberia, the decomposition of logging slash comprised an insignificant proportion of carbon flux to the atmosphere compared to fire emissions and post-fire decomposition (Vedrova et al., 2002). It should also be noted that, unlike natural disturbance, harvesting tends to be concentrated on the most productive portions of the landscape. This could give it an impact out of proportion to area affected (Li et al., 2003).

For a more complete discussion of forest management's impacts on the carbon budget, see chapter 10, this volume.

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Across the Canadian boreal and north temperate forests, insects and pathogens annually cause forest volume losses through mortality and growth reductions that are three times the volume lost to fire.

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## CLIMATE CHANGE IMPACTS ON BOREAL CARBON DYNAMICS

The most pressing question is how climate change will affect the carbon balance in the boreal forest. A warming climate could change the productivity/respiration



balance, change disturbance regimes, shift forest types, and possibly cause dramatic changes in the extent of the biome itself.

### Increased productivity versus increased respiration

Much of the uncertainty regarding carbon flux under a changing climate revolves around whether rates of respiration (both autotrophic and heterotrophic) will increase faster than rates of photosynthesis. There is also a question of whether such increased rates will be sustained, or will only constitute a short-term reaction.

If climate change results in warmer temperatures in early spring, forest productivity could respond positively thanks to the extension of the growing season (Chen et al., 1999). This could have the greatest effect in deciduous forests due to the stronger response to early-season warmth (Barr et al., 2007). On the other hand, if rising spring temperatures are erratic, they could cause growth reductions by stimulating early de-hardening of tree buds which are then susceptible to frost damage (Hanninen et al., 2005). If rising temperatures come later in the growing season, when moisture stress is a potential problem, then either growth increases could be outstripped by respiration increases (Lindroth et al., 1998), or photosynthesis could actually decrease (Kang et al., 2006). For example, 20<sup>th</sup> century decreases in white spruce growth in Alaska have been linked to increased drought stress caused by rising temperatures (Barber et al., 2000). The most common response of trees at the northern Alaskan treeline to increasing temperature is growth reduction, especially on productive sites where competition for moisture is high (Wilmking et al., 2004). Exclusion of such drought impacts from boreal models could potentially skew projections of the carbon budget (Briffa et al., 1998).

Satellite monitoring of boreal forests reveals that productivity declines may be occurring in some regions, perhaps attributed to moisture stress. Goetz et al. (2007) found that more than 25% of boreal forests in Canada that were not recently disturbed showed a decline in productivity with rising global temperatures. Large areas of Siberia showed increased productivity, but this is likely the result of rigorous post-fire regrowth in the wake of many extreme fire seasons.

Thus, whether or not precipitation rises along with temperature has very important consequences for carbon flux (Pastor and Post, 1988). If temperature and precipitation increase in tandem, Fennoscandian forests may demonstrate increased productivity (Kellomaki et al., 1997). Predictions of future precipitation changes show strong variation across the boreal system, and even within select ecozones. For instance, while precipitation is expected to increase across most of northern Europe, it is forecasted to decrease in southern Fennoscandia (Flannigan et al., 1998). Similarly, while increased drought stress is modeled for interior Canadian forests, precipitation could rise in maritime eastern Canada (Amiro et al., 2001).

Changing temperature and precipitation regimes will affect decomposition rates in the future. Increasing soil temperatures could increase mineralization and breakdown of organic matter, potentially making more nutrients available for tree growth (Van Cleve et al., 1990). However, the supply of labile nitrogen in the soil may

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be depleted fairly quickly. In addition, any nitrogen-induced increases may be outweighed by concomitant increases in soil respiration (Bonan and Van Cleve, 1992). Soil respiration may be particularly important if a greater proportion of the increased growth goes into roots than aboveground structures (Niinisto et al., 2004).

However, it is heterotrophic respiration that holds the greatest potential for turning boreal forests from sinks to sources in a warming climate. Bonan and Van Cleve (1992), using models that simulated production and decomposition under warming conditions in Canadian forests, found that respiration increases would balance out photosynthesis gains in black spruce and paper birch (*Betula papyrifera*) forests, and would exceed them in white spruce forests. In a simulation of climatic warming in Finland, gross primary production increased by 12%, but respiration by 22% (Makipaa et al., 1999). However, climatic simulation in Alaska predicted that increases in heterotrophic respiration would only exceed productivity increases in paper birch stands, while the opposite would be true in white spruce and balsam poplar (*Populus balsamifera*) stands (Yarie and Billings, 2002).

Experimental soil warming (+5° C) in north-temperate forests in Maine increased respiration by 25-50% (Rustad and Fernandez, 1998). Much of the increase could come from decomposition of deep soil carbon, which currently comprises a small proportion of the whole (Winston et al., 1997; Goulden et al., 1998). In Siberian forests with extreme buildup of organic matter, warming conditions could cause long-term, sustained increases in heterotrophic respiration from humified materials (Dioumeava et al., 2002). Increased heterotrophic respiration may be limited by certain factors, however. Since the amount of labile organic matter is limited in many boreal soils, respiration rates may tail off after this pool is “burned off” by increased decomposition, (Rustad and Fernandez, 1998). In addition, microbial communities in the soil may acclimate to higher temperatures, regulating decomposition rates (Jarvis and Linder, 2000; Bronson et al., 2008).

The potential for increases in deep soil decomposition is greatly increased if significant soil thawing and permafrost degradation occurs. This will largely be determined by how a changing climate affects the litter and bryophyte layers that insulate the soil profile. Increasing fire in a warming climate could reduce the thickness of these insulating layers (Harden et al., 2000), and warmer air temperatures would increase the period of time in which there is a positive heat flow from the atmosphere to the ground layer (Kasischke et al., 1995). Both of these factors could cause degradation of permafrost. Camill (2005) found that increasing air temperatures in the latter half of the 20<sup>th</sup> century (without an accompanying increase in precipitation) resulted in widespread degradation across the discontinuous permafrost zone of Manitoba. However, drying of the litter layer could reduce decomposition rates (Niinisto et al., 2004), and reduce the layer’s thermal conductivity, thereby decreasing the depth of soil thawing (Bonan et al., 1990). If precipitation were to increase along with temperature, this drying would be prevented and permafrost thaw could increase (Gorham, 1991).

The impact of changing temperatures and precipitation is especially hard to understand in boreal peatland systems. On one hand, permafrost degradation and

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increased heterotrophic respiration are significant possibilities (Hobbie et al., 2000). On the other hand, peat accumulates twice as fast on “collapse scars” as on bogs with intact permafrost (Camill et al., 2001). Thus, the increased productivity of these areas could offset some of the carbon losses. There is also a tradeoff in peatlands between aerobic decomposition (which releases CO<sub>2</sub>) and anaerobic decomposition (which releases CH<sub>4</sub>). If water tables drop, aerobic decomposition is likely to increase, since waterlogged peat is oxygen-poor, but affected areas could also experience reductions in CH<sub>4</sub> emissions as anaerobic decomposition declines. Under this scenario, it is unclear whether peatlands will become a source or sink. Dried-out peatlands will have accelerated oxidation of organic matter, but reduced emissions of CH<sub>4</sub>, whereas waterlogged, collapsed thermokarst basins will accumulate more peat resulting in increased CH<sub>4</sub> emissions (Gorham, 1991).

### **Changing disturbance regimes**

Cycles of forest fire and insect outbreak are controlled by weather and the condition of the fuel or host. Both of these factors could be altered by climate change. One possibility is a more rapid build-up of pandemic insect populations as increasing temperatures could cause drought stress in their host tree species as well as shorten insect life cycles. A massive spruce beetle outbreak in Alaska has been attributed to abnormally warm and dry summers since the 1960s (Berg et al., 2006), and similar climatic triggers may be causing the widespread devastation by mountain pine beetle across western North America (Malstrom and Raffa, 2000; Powell and Logan, 2005). Indeed, the prospect of future pine beetle and spruce budworm outbreaks caused one model to predict that Canadian boreal forests will be a net source of greenhouse gases in the coming decades (Kurz et al., 2008).

Climate change may also allow pests that are less cold tolerant to extend their distribution into the boreal zone (Wolf et al., 2008). However, it may also be possible that a warming climate could suppress insect populations under certain conditions. One model predicts that rising temperature without an accompanying rise in precipitation will decrease the area affected by spruce budworm in temperate forests of Oregon (Williams and Liebhold, 1995).

There is evidence that fire return intervals have been shortening across the boreal forest during the 20<sup>th</sup> century, and this trend could continue (Stocks et al., 1998). Annual area of North American boreal forests burned increased approximately by a factor of three between the 1960s and the 1990s (Kang et al., 2006). One study predicted that Canadian fire return intervals could decline from an average of 150 years to 100-125 years, with significant associated emissions (Kasischke et al., 1995). And just as future rates of photosynthesis and respiration will depend on how precipitation changes in relation to rising temperatures, so too will future fire return intervals (Flannigan et al., 1998; Amiro et al., 2001). It is possible that the most significant impact of rising CO<sub>2</sub> levels in the atmosphere thus far has been an increase in fire frequency, thus altering the boreal forest age-class distribution (Bond-Lamberty et al., 2007).

### Changes in biome and forest type

Some research predicts significant compositional changes within the boreal zone with a changing climate, as well as a shift of its southern border northward with expansion of temperate forests and steppe and invasion of its northern border into the tundra. Some predictions are dramatic: Emanuel et al. (1985) modeled that boreal forests will decrease by 37% if there is a doubling of atmospheric CO<sub>2</sub> concentration. Rising temperatures and degrading permafrost are allowing Siberian kedar (*Pinus sibirica*) to invade the understory of larch stands across southern Siberia and Mongolia, and coniferous forests are displacing montane tundra in the mountain ranges of these regions (Soja et al., 2007). In boreal Canada, climate change may make deciduous forest types more competitive (Kasischke et al., 1995), perhaps due to increased fire that favors the hardwood pioneers birch and aspen. A shift to hardwood dominance could change future fire regimes, nutrient dynamics, and even the boreal climate, since the albedo of deciduous forests is higher than coniferous types (Amiro et al., 2006; Goetz et al., 2007). However, caution should be used in predicting major compositional changes through modeling. Models are convenient for parametrizing and testing assumptions about complex questions, but the results are only as good as the available data, the assumptions used, and the ability to calibrate and verify the model. Data on feedback between climate and boreal forests are very limited and highly variable, leading to highly variable model results. For example, one model in Alaska predicted that moisture-induced stress would cause the disappearance of existing forest types and their replacement by aspen woodlands (Bonan et al., 1990), but later refinement of the model to include more parameters of biophysical complexity indicated that moisture deficits would likely not reach levels that could cause such widespread mortality (Bonan and Van Cleve, 1992).

Compositional changes within the boreal zone could significantly alter carbon dynamics, but conversion of boreal forests to temperate forests, or tundra to boreal forests, could have a greater impact. Such transitions will not be rapid. Rather, the existing community will likely degrade at a faster rate than new vegetation types can invade. During the lag, large CO<sub>2</sub> emissions are possible (Apps et al., 1993). Smith and Shugart (1993) predicted a net carbon loss of 36.6 Pg over a 50-100 year period as other forest types invade the boreal region. The movement of boreal forests into the tundra could greatly increase fuel loads, bringing fire into a system in which it is rare (Kasischke et al., 1995). The impact on soil carbon pools in the tundra is unknown, but concerning. In addition, northward migration of the tree line will change albedo levels in high northern latitudes.

### Albedo effect

Albedo is not directly related to carbon storage and release; rather, it controls the absorption of heat by the biome. At high northern latitudes, forest cover increases heat absorption because dark conifer crowns have lower albedo (less reflectivity) than low, snow-covered tundra vegetation. The result is that boreal forests may actually exert a warming influence on regional and global climate, which may outweigh their potential role as carbon sinks (Betts, 2000). The presently high albedo of tundra

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It is possible that the most significant impact of rising CO<sub>2</sub> levels in the atmosphere thus far has been an increase in fire frequency, thus altering the boreal forest age-class distribution.

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creates a feedback with the Arctic Ocean, maintaining high levels of sea ice; forest invasion of the tundra zone could alter this interaction, changing dynamics across the entire polar region (Bonan et al., 1995). One modeling exercise that replaced global boreal forests with grass and shrub vegetation predicted a cooling of the earth's climate because of the greater reflectance of these vegetation types (Bala et al., 2007). This research suggests that albedo effects may have a dominant influence on climate at high latitudes. It should be considered, however, that these conclusions are heavily reliant on modeling, and are a relatively recent addition to boreal zone research. At the very least, however, the albedo effect should be considered as a potential balance to any effect that boreal forests may have on slowing climate change through carbon sequestration.

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The presently high albedo of tundra creates a feedback with the Arctic Ocean, maintaining high levels of sea ice; forest invasion of the tundra zone could alter this interaction, changing dynamics across the entire polar region.

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### CONCLUSIONS

Much of the research regarding the impacts of climate change on the boreal carbon budget is based on modeling, and can only predict potential changes. However, some observations of existing impacts are available, and seem to point toward the potential for greater carbon loss from boreal forests. Steadily increasing temperatures across boreal and arctic North America in the past fifty years have been associated with drought-induced growth reductions, permafrost degradation, increased fire frequency, increased soil respiration, and potentially larger outbreaks of insect pests. Under these conditions, increased respiration associated with rising temperatures seems to outstrip any increases in carbon uptake through growth. The possibility of greatly altered carbon dynamics due to permafrost degradation also exists.

However, there is also research suggesting that some of the impacts of climate change may not be as extreme as predicted. It is unclear whether increased soil temperatures will cause a sustained increase in carbon release. The pool of labile carbon in the soil may not be large, resulting in only a brief increase in decomposition. While the degradation of permafrost may increase the release of CO<sub>2</sub>, it could also result in reduced emissions of CH<sub>4</sub>, a potent greenhouse gas. Some models also predict an increase in precipitation across much of the boreal zone, which in concert with rising temperatures could cause increased productivity.

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## Chapter 7

# Methods of Measuring Carbon in Forests

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### EXECUTIVE SUMMARY

Accurate measurement of carbon stocks and flux in forests is one of the most important scientific bases for successful climate and carbon policy implementation. A measurement framework for monitoring carbon storage and emissions from forests should provide the core tool to qualify country and project level commitments under the United Nations Framework Convention on Climate Change, and to monitor the implementation of the Kyoto Protocol.

Currently, there are several methods for estimating forest carbon stocks and flux, ranging from the relatively simple forest biomass inventory to complex, sophisticated experiments and models. Advanced carbon estimation methodologies such as LiDAR and eddy covariance carbon flux experiments may provide reliable, accurate and transparent data and serve as a basis for market tools and international policymaking such as carbon trading, carbon taxes, and credits for reducing emissions from deforestation and forest degradation in developing countries (REDD, REDD+). Nevertheless, developing countries, which have limited capacity for data collection and management, need low-cost methodologies with acceptable spatial and temporal resolution and appropriate sampling intensity.

If a standardized verification system across projects, countries, and regions is to ever be attained, policymakers should be aware that there are different basic approaches to measuring forest carbon, which have advantages and disadvantages, and varying degrees of accuracy and precision.

We review the four categories of methods for measuring forest biomass and estimating carbon which are currently in use: i) forest inventory (biomass); ii) remote sensing (relationship between biomass and land cover); iii) eddy covariance (direct

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measurement of CO<sub>2</sub> release and uptake); and iv) the inverse method (relationship among biomass, CO<sub>2</sub> flux and CO<sub>2</sub> atmospheric transport). These methods all vary in their level of accuracy and the resolution at which data can be obtained. Each technique has its own advantages and disadvantages and there are appropriate circumstances for using each one in measuring CO<sub>2</sub> flux and carbon storage for different temporal and spatial scales of evaluation and measurement.

Forest inventory methods are direct measures of biomass accumulation within a forest.

They have a long history in development and good data is generally available; however, they are low in time resolution, costly to implement, require technical training and knowledge, are variable in standards for measurement, and are available in only certain regions, mostly developed countries.

Remote sensing methods usually are combined with models that link remote sensing information with CO<sub>2</sub> and carbon data (often forest inventory information). Methods can be divided into passive sensing (satellite images, aerial photographs that are characterized by reflected light) and active sensing (radar, LiDAR that emit and receive microwaves or light respectively). Remote sensing is limited by incomplete information, resolution and detection problems, and uncertainties in models that require further development and refinement. Nevertheless, when available at a suitable resolution and spatial scale, it can be the cheapest method of surveying forests.

The eddy covariance method is advanced in its accuracy and resolution, and is a good method for direct measurement of small (hectare-plus) scale CO<sub>2</sub> flux; but, it is still restricted by systematic biases, is not accurate in rough topography, and has limited observation sites around the world.

Inverse methods typically are used at continental or global scales. These methods calculate the total sources and sinks, including both anthropogenic and natural, using available atmospheric CO<sub>2</sub> concentration data and transportation models. Carbon Tracker is one of the most advanced inverse methods. It was developed by NOAA's Earth Systems Research Laboratory as a system to keep track of carbon dioxide uptake and release at the Earth's surface over time and to continuously improve models and data assimilation methods for higher accuracy and resolution.

### **What we do and do not know about measuring carbon in forests**

- Forest inventory methods require historical and regional data. Permanent continuous forest inventory (CFI) plots are the best to provide long-term accurate and non-biased assessments. Non-permanent plots can be used but are often biased.
- Most developed countries conduct regular national inventories to evaluate forest health and status. These inventories are therefore a useful data base if biases can be avoided.
- In the past, inventory plots have often been biased toward sampling forests of commercial value. Forests considered degraded or that are now growing back (secondary forest) are often under-represented. Inventories often only

include tree species that have commercial value and under-sample small trees.

- Very few inventories account for belowground biomass, litter, and dead wood. Fine spatial-resolution (1-10 m) satellite data have the advantage in providing high resolution details of a specific area. However, disadvantages include a small area of coverage, shadows, and expense in acquisition.
- It is expensive to sample a sufficient number of trees representing the diversity of size and species to generate local allometric equations for use in converting tree data to forest biomass data.
- Medium spatial-resolution (10-100 m) satellite data are the most suitable for regional level above-ground biomass estimation because of better data availability (spatial and temporal), and the lower cost of acquisition and storage. Since spatial resolution is usually sufficient to compare with inventory measurements, this approach is widely used for forests.
- Coarse spatial resolution satellite data (> 100 m) are most effective at large national or continental scales. The use at such scales is limited, however, because of the occurrence of mixed pixels, and differences between scale and resolution of forest inventory measurements.
- Aboveground biomass estimation by radar can achieve good accuracy in low and medium density forests, but the relationship between radar backscatter and aboveground biomass weakens when the forest becomes too dense. Its advantage is its ability to penetrate precipitation and cloud cover, and avoid shade/shadow effects from the sun.
- Light Detection and Ranging (LiDAR) is an active remote sensing method, analogous to radar, but using laser light instead of microwaves. The technology needs further development to be widely useful in aboveground biomass estimation.
- Recent technical, financial and logistical (scheduling) problems with the U.S. remote sensing program highlight the need for more countries or consortiums to provide the international remote sensing community with more options in satellite imagery and Radar/LiDAR data.
- Eddy covariance measurements have been continuously made at certain sites for over ten years. New observation sites (especially in tropical forest regions), updated models, and remote sensing data will enable eddy covariance methods to continually refine estimates of CO<sub>2</sub> flux from regional to continental scales, making eddy covariance the world's direct tracking system of carbon flux.
- More research needs to be conducted to close the energy budget in eddy covariance measurements and eliminate biases caused by nighttime stratification and complex topography.



- CarbonTracker has emerged as one of the most advanced inverse models currently used for regional and continental inverse estimates of carbon sinks and sources.

**Keywords:** biometrics, carbon flux, Carbon Tracker, climate change, eddy covariance, forest inventory, global observation network, inverse methods, remote sensing, sequestration

## INTRODUCTION

The need to accurately measure the stocks and flux of carbon in forests is urgent given the global consensus that CO<sub>2</sub> emissions have a very strong influence on global warming. Forests are an essential part of the carbon cycle. They are a major terrestrial sink of CO<sub>2</sub>, but their land use conversion to agriculture currently accounts for 25% of global carbon emissions. Compared to the combustion of fossil fuel, emissions from land use change are an important issue for developing countries and especially for tropical countries (Houghton and Ramakrishna, 1999). Forests are influenced by various anthropogenic and natural disturbances such as fire, disease, insect infestations, harvesting, deforestation, and degradation, all of which can lead to significant carbon emissions. To understand the carbon cycle in the forest, it is important to have valid, cost-effective scientific methods to measure and monitor carbon. Such measures require accuracy and precision in order to have useful data on carbon stocks and flux in forests globally.

Accurate estimation of forest carbon stocks and flux in is one of the most important scientific bases for successful policy implementation. Although understanding the methods of measuring the forest carbon cycle may not be a focus of policymakers, it is important that they recognize that there are differences between regions and countries in carbon emission behaviors and carbon storage in forests (and associated land conversion). This understanding will allow them to make better decisions about global and regional resource allocation for measurement capacity, and therefore to optimize adaptation and mitigation strategies for climate change. A measurement framework for monitoring carbon storage and emissions from forests should be the core tool to qualify country and project level commitments under the United Nations Framework Convention on Climate Change (UNFCCC, 1997), and to monitor the implementation of the Kyoto Protocol (Brown, 2002).

To meet the requirements of the Kyoto Protocol, all Annex I countries<sup>1</sup> must “provide data to establish their level of carbon stocks in 1990 and to enable an estimation of its changes in carbon stocks in subsequent years” (UNFCCC, 1997). Developing countries, which have limited capacity in data collection and management, need methodologies with low-cost, acceptable spatial and temporal resolution and appropriate sampling intensity. Furthermore, for the post-Kyoto era, advanced carbon estimation methodologies may provide reliable, accurate, and transparent data and serve as a basis for market tools and international policymaking

<sup>1</sup> Annex I Parties to the United Nations Framework Convention on Climate Change (UNFCCC) include the industrialized countries that were members of the OECD (Organisation for Economic Co-operation and Development) in 1992, plus countries with economies in transition (the EIT Parties), including the Russian Federation, the Baltic States, and several Central and Eastern European States.

such as carbon trading, carbon taxes, and credits for reducing emissions from deforestation and forest degradation in developing countries (REDD, REDD+).

### Objectives

In this chapter we describe four basic methods of measuring carbon storage and flux in forests: i) forest inventory; ii) remote sensing; iii) eddy covariance; and iv) the inverse method. These methods are critiqued for their advantages and disadvantages in estimating CO<sub>2</sub> flux and storage. All are evaluated for their accuracy and resolution. In the conclusion section, we describe gaps in data, information, and technologies that need to be addressed if a standardized measurement framework is to be achieved. Recommendations are made on improvements in methodology for more efficient and effective aboveground biomass (AGB) estimation.

### Measuring carbon

Generally, there are two main approaches to measuring carbon stocks and fluxes in each forest carbon pool: (i) measuring changes in carbon stock, and then inferring a carbon flux under a certain level of confidence; and (ii) measuring carbon flux directly. Generally, biomass, which is readily measured, is widely used to estimate carbon stocks using proven formulas for the ratio of carbon to biomass instead of measuring carbon directly, particularly for aboveground carbon (Brown, 1997).

Carbon stocks in forests can be classified into five different measurement pools:

- *Aboveground biomass* – Living biomass above the soil, including stem, stump, branches, bark, seeds, and foliage. This category includes live understory.
- *Belowground biomass* – All living biomass of roots greater than a certain diameter.
- *Dead wood* – Includes all non-living woody biomass either standing, lying on the ground (but not including litter), or in the soil.
- *Litter* – Includes the litter, humus layers of the soil surface, and all non-living biomass of a certain diameter lying on the ground.
- *Soil organic carbon (SOC)* – Typically includes all organic material in soil to a depth of 1 meter, excluding the coarse roots of the belowground biomass pool.

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Developing countries, which have limited capacity in data collection and management, need methodologies with low-cost, acceptable spatial and temporal resolution and appropriate sampling intensity.

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### FOREST INVENTORIES AND ABOVEGROUND CARBON STOCK ESTIMATIONS

Because national forest inventories are commonly available for many countries, different approaches have been developed to estimate above ground biomass (AGB) from inventories. They can be categorized by data source: (i) field measurement; (ii) remote-sensing data; or (iii) ancillary data used in GIS-based modeling (Lu, 2006; Wulder et al., 2008). Several approaches to estimating carbon stocks from each of these data sources are shown in Table 1.

**Table 1 Summary of techniques for above ground carbon stock estimation**

<i>Category</i>	<i>Methods</i>	<i>Data used</i>	<i>Characteristics</i>	<i>References</i>
Field measurement methods	Conversion from volume to biomass by biomass expansion factor (BEF)	Volume from sample trees or stands	Individual trees or vegetation stands	Fang et al., 2001 Smith and Heath, 2004 Wang et al., 2007a Woodbury et al., 2007 Wulder et al., 2008
	Allometric equations	Sample trees	Individual trees	Gehring et al., 2004 Goodale et al., 2002 Jenkins et al., 2003 Zianis and Mencuccini, 2004
Remote sensing methods	Methods based on fine spatial-resolution data	Aerial photographs, IKONOS	Per-pixel level	Thenkabail, 2003 Thenkabail et al., 2004a
	Methods based on medium spatial-resolution data	Landsat TM/ETM + , SPOT	Per-pixel level	(Dong et al., 2003) Muukkonen and Heiskanen, 2005 Muukkonen and Heiskanen, 2007 Cohen and Goward, 2004 Lu and Batistella, 2005
	Methods based on coarse spatial-resolution data	IRS-1C WiFS, AVHRR	Per-pixel level	Cross et al., 1991 Laporte et al., 1995
	Methods based on radar data	Radar	Per-pixel level	Blackburn and Steele, 1999 Levesque and King, 2003 Sun et al., 2002
	Methods based on LiDAR Data	LiDAR	Per-pixel level	Anderson et al., 2006 Drake et al., 2003 Lefsky et al., 1999

Source: Modified from Lu (2006)

### Field-based methods

The field-based method is usually referred to as an inventory assessment, and can be further classified into volume-to-biomass and diameter-to-biomass approaches. The choice between these approaches is dependent upon the data available and the desired resolution. Generally, the approach of converting timber volume, which is commonly

available, to biomass has more uncertainty but requires less detailed data; therefore, this is the most commonly used method. If detailed diameter information and field measurements are available for establishing allometric equations, then the diameter-to-biomass (allometric) approach is generally favored because it is more accurate.

Timber volume data are available for many countries because these data are primarily collected for forest management and revenue accounting. In 1919 (Norway), 1921-24 (Finland), and 1923-24 (Sweden), the Nordic nations started national forest inventories because of the fear that the fuelwood resource would be exhausted (FAO, 2000; Brack, 2009). Optimally, species, diameter at breast height (DBH), height, site quality, age, increment, and defects are recorded in each inventory dataset (LeBlanc, 2009). However, different countries have various capacities and standards for detailing the inventory information. For example, *Forest Statistics of China 1984-1988* is compiled from more than 250,000 permanent and temporary plots across China, and the technical standard in data collection includes measuring DBH, height, stem volume, age, total area, and site quality (Fang et al., 1998). But in the *National Forest Inventory of Indonesia 1989-1996*, only the number of trees per ha and volume per ha for different diameter classes is available (FAO, 2000). In Brazil, very limited data collection is done regionally by consultants, but not by the government or the research academy (Freitas, 2006; Wardoyo, 2008). It is therefore necessary for some countries to utilize available timber volume data from private company and landowner inventories so as to obtain rudimentary baseline domestic estimates of changes and stocks of standing forest carbon.

#### *Estimating biomass from timber volume*

The biomass expansion factor (BEF) is defined as the ratio of all standing aboveground biomass (AGB) to growing stock volume ( $\text{Mg}/\text{m}^3$ ) (Fang et al., 2001). It has been developed to estimate aboveground biomass when timber volumes within diameter classes are reported (Brown, 2002). Especially for estimating large areas within developing countries that lack detailed information about forest biomass, the BEF is a practical estimate of AGB.

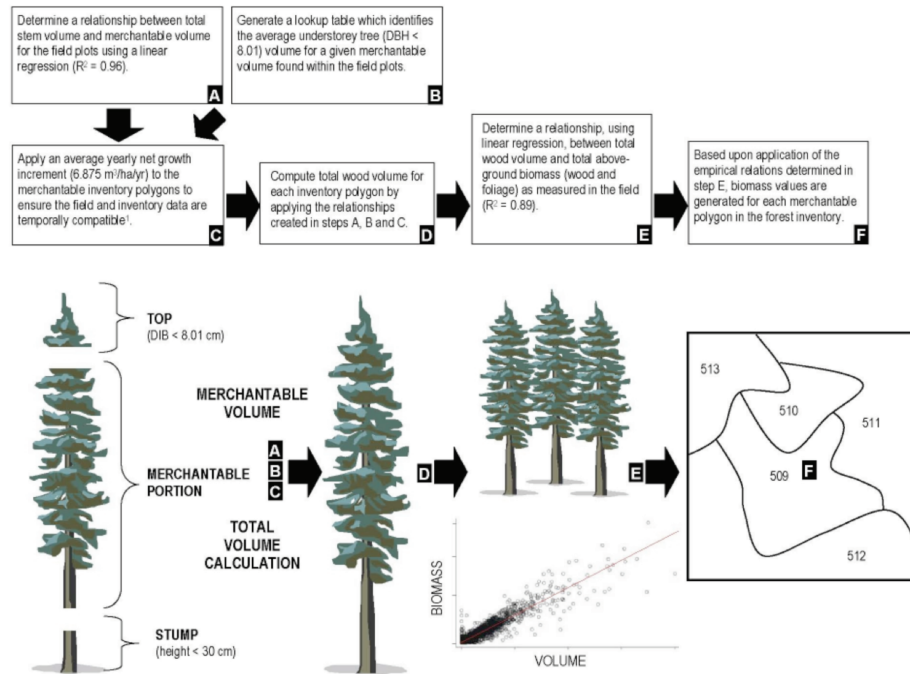
The process of estimating carbon stock by BEF can be simply to use the regression relationships between merchantable plot tree volumes, their annual increments, and estimates of non merchantable volumes, to above ground standing biomass. Estimations of total aboveground biomass from tree volume data is then subsequently expanded to an area based on uniformity of site, stocking and age-class distribution (see Figure 1 for example) (Wulder et al., 2008). BEF varies by different stand density-related factors, such as forest age, site class, stand density, and other biotic and abiotic factors (Brown et al., 1999; Fang et al., 2001). The largest differences are regional and by forest type (see Figure 2) (Brown, 2002).

#### *Estimating biomass from tree diameter*

Compared to the BEF method, allometric equations can provide more precise estimates of aboveground biomass. In the biological sciences, the study of the

relationship between the size and shape of organisms is called allometry (Niklas, 1994). In the context of biomass estimation, allometry refers to the relationship between individual tree diameters (sometimes with heights) and aboveground biomass for specific species, groups of species, or growth form (Jenkins et al., 2003; (Zianis and Mencuccini, 2004).

**Figure 1** An overview of the process used to estimated biomass from the forest inventory data

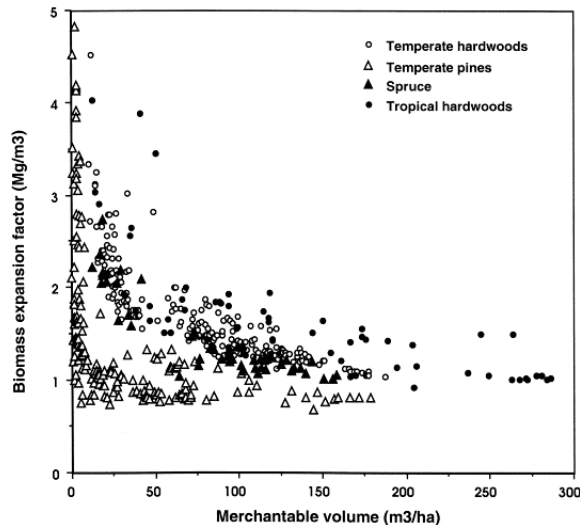


<sup>1</sup>After calculating the merchantable volume of each study plot, the difference between each calculated merchantable volume and the corresponding merchantable volume from the forest inventory database was found, and divided by the number of years between the two data sets. The average yearly difference, or net growth increment, of all study plots 6.875 m<sup>3</sup>/ha/yr was applied to the merchantable volume of the forest inventory polygons, creating a dataset which was now compatible with the field data.

Source: From Wulder, M.A., White, J.C., Fournier, R.A., Luther, J.E., Magnussen, S., 2008. Spatially explicit large area biomass estimation: Three approaches using forest inventory and remotely sensed imagery in a GIS. *Sensors* 8, 529-560. Reprinted with permission.

In order to derive an accurate allometric equation for any forest type, an adequate sample of tree sizes and species must be taken. If such data are available at the appropriate scale, the allometric approach can be very accurate. Generally, species groups such as tropical wet-evergreen hardwoods, temperate eastern U.S. hardwoods, pines, and spruces produce highly significant correlations of greater than 0.98 for regressions between diameter at breast height (dbh) and biomass per tree (Brown, 1997; Schroeder et al., 1997; Brown et al., 1999; Brown, 2002). A study on in lianas in Amazon semi-evergreen rain forest showed that a combination of diameter and length is also significantly correlated with biomass ( $R^2 = 0.91$ ) (Gehring et al., 2004). This approach is limited, however, by the lack of allometric data for many forest types and regions.

Figure 2 Relationship between BEF for temperate hardwoods, pines and spruce, and tropical hardwoods



Source: From Brown, S., 2002. Measuring carbon in forests: current status and future challenges. *Environmental Pollution* 116, 363-372. Reprinted with permission.

### *Improvement for field based methods*

Estimates of carbon flux from forest inventory measurements require availability of historical data at the regional scale. All developed countries conduct regular national inventories (FAO, 2000). For the 137 developing countries, 22 have repeated inventories, 54 have a single inventory, 33 have partial forest inventories, and 28 countries have no inventory (Holmgren and Persson, 2002). In the U.S., a vast network of permanent sample plots makes up the Forest Inventory and Analysis (FIA) and Forest Health Monitoring (FHM) programs. The FIA program, which has been operating for about 70 years, periodically measures all plots on a state-by-state basis every 5-14 years (Brown, 2002; Smith et al., 2002).

Inventory data have several deficiencies that can bring uncertainty, however. First, inventories tend to be conducted in forests that are considered to have commercial value, and the forests that many people depend upon for other values (such as water, recreation, open space, or subsistence) may not be included. Many degraded or semi-deforested open lands, or those regions that are now growing back (secondary forest) are under-sampled or not measured. Often only trees species that have commercial value at the time of the inventory are counted (Brown, 1997). This counting bias can bring systematic inaccuracy to the estimation of carbon. Additionally, the assumption that small trees (about 10 cm diameter or less) contribute little to the total forest biomass is not robust according to Schroeder et al. (1997). They concluded that for young hardwood stands in the eastern USA with aboveground biomass less than 50 Mg/ha, trees with dbh of 10 cm or less contain as much as 75% of the biomass of trees with dbh greater than 10 cm.

The cost is high to sample a sufficient number of trees representing a range of size and species in order to generate local allometric equations (Brown, 2002). Many developing countries lack funding, staff, and expertise to acquire the data. Additionally, a small number of large diameter trees (>100 cm) and a large number of small diameter trees (<10 cm), which are important to the total biomass, are often missed in a sample for allometry measurements (Brown, 1997).

To improve the accuracy and precision of measuring aboveground live tree biomass by inventory methods, Brown (2002) has suggested that the following:

- Destructively harvest large diameter trees to establish allometry equations, because they are under-sampled and they have a significant influence on the regression relationship between diameter and biomass.
- Precisely measure small trees (10 cm diameter or less) for temperate hardwood forests (i.e. second growth) or other forest types in which small diameter trees may be significantly underestimated.
- Including height in regression equations can slightly improve the precision, but given the difficulty of measurement, it is not feasible or worth the effort for large areas. The use of remote sensing data can complement tree height data for large-areas, and can improve the precision of allometric regression equations.
- Periodically re-visit the field sites from which the inventory data are derived and modify the allometric equations that may have changed with time and forest growth.

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#### REMOTE SENSING METHODS

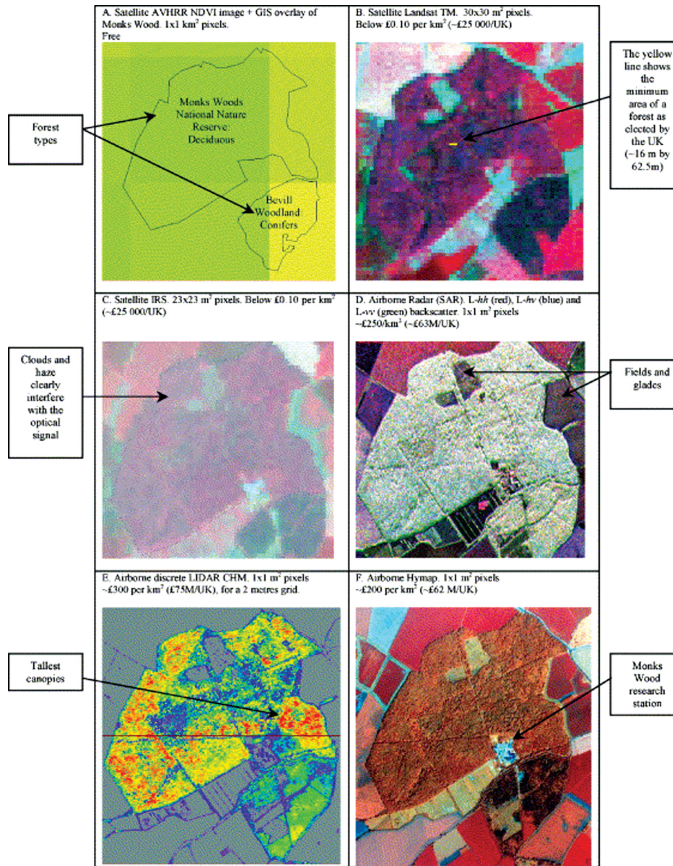
Inventory data have been used as the basic approach to estimating carbon stock in existing and historical forests worldwide. In recent years, better models and the establishment of more plots have improved accuracy and precision (Smith and Heath, 2004). However, sampling intervals are long (5-14 years), so temporal resolution of changes in carbon storage is limited. In addition, gathering inventory data is highly dependent on the capacity of local people to conduct the survey. Assuming that land use change accounts for a significant part of carbon emissions, and that the rate of deforestation is high, remote sensing would appear to be a more suitable method, particularly for use in large and remote forest regions and in developing countries where training on forest inventory procedures is poor.

The remote sensing method monitors forests at different temporal, spatial and spectral resolutions (Patenaude et al., 2005). Several applications of remote sensing for mapping land covers are available and can be categorized as passive (optical) or active (radar).

Optical, or passive, remote sensing technologies include aerial photographs of various kinds (infrared, color, black and white), Normalized Difference Vegetation Index (NDVI) images that are derived from an advanced very high resolution radiometer (AVHRR) sensor, and images from Landsat Thematic Mapper (TM) false

color composites and its associates that are at a low resolution (Figure 3). Active remote sensing technologies include radar and LiDAR derived images. These can measure structure, detect objects below canopy, and can depict canopy height and stratification (CHM) (Figure 3).

Figure 3 Example of different remote sensing methods on the same site.



(A) NDVI image derived from images collected over a 10-day period by the advanced very high resolution radiometer (AVHRR) sensor. A GIS overlay shows the boundaries of the woodlands. (B) Landsat 1989 TM false color composite image. In a false color image, green vegetation appears red to the human eye. (C) May 1998 false color composite image from the Indian Remote Sensing Satellite (IRS). A vector layer has been added to help visualize the boundaries of the woodlands. Note that the atmospheric conditions (haze) clearly affect the quality of the image. (D) June 2000 SAR image from the German Aerospace Research Establishment (DLR). The images show an L-hh, hn, nn backscatter image. The woodland is not fully covered by the imagery available. (E) Canopy heights (CHM) derived from LiDAR discrete data collected in July 2000. Blue areas indicate canopy heights smaller than 10 m; green, larger than or equal to 10 m but smaller than 18 m; red, larger than or equal to 18 m. (F) Airborne hyperspectral data false color image (green vegetation shown as red).

Source: From Patenaude, G., Milne, R., Dawson, T.P., 2005. Synthesis of remote sensing approaches for forest carbon estimation: reporting to the Kyoto Protocol. *Environmental Science & Policy* 8, 161-178. Reprinted with permission.

**Optical remote sensing**

Optical remote sensing captures solar energy reflected by the forest canopy in the visible, near, and middle infrared portion (0.4 to 2.5 mm) (Patenaude et al., 2005). Optical remote sensing is also called passive remote sensing and can be differentiated from Radar and LiDAR methods, which actively emit radiation and then detect the reflectance. The ground sampling distance (GSD) defines the spatial resolution level of the optical remote sensing methods. It can be classified based on degree of resolution into fine, medium, and coarse spatial scales.



### *Fine spatial-resolution data*

Fine spatial-resolution data has a GSD less than 10 m. Aerial photographs (GSD 1.00 m), IKONOS (GSD 0.83 m), and QuickBird (GSD 0.61 m) images are the commonly available fine spatial-resolution data (Lu, 2006).

Aerial photographs were widely used in forest surveys starting in the late 1940s, primarily for forest type delineation and stratification, and timber volume estimation (Lu, 2006). Since the 1990s, space-borne high spatial-resolution satellite images can also be used in biomass estimation as well as in detecting biophysical parameters (height, classification, stand structure). Such images can be used to detect the structural diversity of a forest at a small scale. For example, the IKONOS system, started in September 1999, collects panchromatic data, with a spectral range of 450 to 900 nm, and four GSD channels of 4 m resolution multi-spectral data (Wulder et al., 2004). Thenkabail et al. (2004b) used multi-date wet and dry season IKONOS images to calculate carbon stock levels of the West African oil palm plantations. It was also used by Thenkabail (2003) to detect small differences in floristic association in the Central African rainforest.

Fine spatial-resolution remote sensing data has the advantage in providing details of a specific area. However, disadvantages include the small area of coverage, preponderance of shadows, and acquisition expense. Therefore, it should mainly be used in small scale projects that are focused on measuring stand-level characteristics (Thenkabail et al., 2004b). Such fine scale resolution can also be useful for the development of reference data for validation or accuracy assessments of medium and coarse scale remote sensing measurements (Lu, 2006).

### *Medium spatial-resolution data*

Medium spatial-resolution remote sensing images (10 m to 100 m) are the most suitable for regional level aboveground biomass estimation because of better data availability (spatial and temporal), and the lower cost of acquisition and storage. Since spatial resolution is still good enough to compare with inventory measurements, this approach is widely used for aboveground biomass estimation for various forests (Reese et al., 2002; Tomppo et al., 2002; Foody et al., 2003; Zheng et al., 2004; Muukkonen and Heiskanen, 2005, 2007). Landsat Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM+), Multi-Spectral Sensor (MSS), ASTER, AVIRS, and SPOT High Resolution Visible (HRV) are all multispectral sensors commonly used for mapping forest structure and estimating biomass (Muukkonen and Heiskanen, 2005).

Landsat has been the most important data source for mapping and remote sensing interpretation. For more than 30 years it has provided appropriate spatial and spectral resolution to detect and characterize forests at an affordable cost (Cohen and Goward, 2004). Since 1972, the Landsat program has launched seven satellites. With each launch, sensors have been designed for better spatial and spectral resolution. Landsats 1, 2, 3, and 4 have been decommissioned because better satellites are now available or they had reached the end of their working life. However, due to the failure

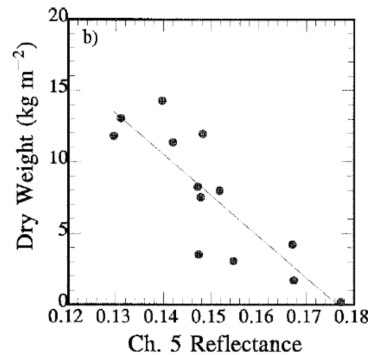
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of Landsat 6 and a defective scan line on Landsat 7, Landsat 5 has been kept running for 24 years and is still widely used for research. The earliest sensor (four-band multi-spectral scanner sensor – MSS) was deployed on Landsat satellites 1 to 5. But because of the lower spatial resolution (80 m), and fewer spectral bands of MSS, the TM instrument, and then later the ETM+ instrument, which have seven spectral bands and 30 m spatial resolution, are now the primary images used in aboveground biomass estimation (Figure 4).

**Figure 4** Aboveground biomass of secondary forest versus TM channel 5 reflectance.



Source: From Steininger, M.K., 2000. Satellite estimation of tropical secondary forest above-ground biomass: data from Brazil and Bolivia. *International Journal of Remote Sensing* 21, 1139-1157. Reprinted with permission.

The Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) was launched in 1999, with three spectral bands in the visible near-infrared region (VNIR), six bands in the shortwave infrared region (SWIR), and five bands in the thermal infrared region (TIR), with 15-, 30-, and 90-m spatial resolution, respectively (Muukkonen and Heiskanen, 2005). In spite of its modernity, it is argued that ASTER has relatively narrow SWIR bands 5-8 which are primarily designed for soil and mineral detection, so it is not particularly sensitive to detecting differences among forests (Yamaguchi et al., 1998).

#### *Coarse spatial-resolution data*

Overall, coarse spatial-resolution data (greater than 100 m) are most effective at large national or continental scales. However, use at such scales is limited because of the frequent occurrence of mixed-landuse pixels (due to the large pixel size), and differences between scale and resolution of forest inventory measurements and image GSD (Lu, 2006). However, the use of fine and medium spatial-resolution data along with coarse spatial-resolution can help estimate aboveground biomass and improve accuracy (Dong et al., 2003; Muukkonen and Heiskanen, 2007; Zheng et al., 2007a).

Commonly used coarse spatial-resolution data include NOAA Advanced Very High Resolution Radiometer (AVHRR), Moderate Resolution Imaging Spectroradiometer (MODIS), and SPOT VEGETATION (Table 2) (Lu, 2006). The AVHRR

has collected over 30 years of data and has often been used to assess large areas of forest cover at the scale of a continent (Iverson et al., 1994). For example, for a 1.42 billion ha region of temperate and boreal forest, Dong et al. (2003) used regression analysis between an NDVI dataset, developed from AVHRR at 8x8 km resolution, over an eighteen year period (1981-1999), and timber volumes from forest inventories to estimate aboveground biomass.

**Table 2 Selected examples of biomass estimation using optical remote sensing data**

Datasets	Study area	Techniques	References
IKONOS	West Africa	Empirical regression	Thenkabail et al., 2004b
Landsat 5	Mauaus, Brazil	Liner and exponential regressions	Steininger, 2000
Landsat 5	Para´ state and Rondonia state, Brazil	Multiple regression analysis	Lu and Batistella, 2005
SPOT VEGETATION	Canada	Multiple regression and artificial neural network	Fraser and Li, 2002
MODIS, ASTER	Finland	Regression models	Muukkonen and Heiskanen, 2007
Aerial Photographs	Suonenjoki, Finland	K nearest-neighbor method and K most similar neighbors	Anttila, 2002
Landsat 5	Sweden	K nearest-neighbor method	Fazakas et al., 1999, Reese et al., 2002
Landsat TM and IRS-1C WiFS	Finland and Sweden	K nearest-neighbor method and nonlinear regression	Tomppo et al., 2002

Source: modified from Lu (2006).

The recent SPOT VEGETATION (VGT) sensor provides imagery with a swath width of 2,250 km and GSD at 1,165 m. Besides the four spectral bands of the SPOT multi-spectral sensor, the Vegetation Instrument has an extra band (0.43 to 0.47  $\mu\text{m}$ ) that is used for the first band (blue) and a 1.65- $\mu\text{m}$  short-wave infrared (SWIR) channel. Fraser and Li (2002) tested the relationship between several values and indexes from VGT and aboveground biomass. The short-wave-based vegetation index (SWVI), in which the SWIR is substituted for the red channels from VGT, has been found to have weak correlation ( $R^2=0.25$ ). The other values (red, NIR, SWIR, and NDVI) have either no relation or poor relation with aboveground biomass, and therefore are not useful.

MODIS is a 36-band spectrometer providing a global dataset every 1-2 days with a 16-day repeat cycle. Bands 1 and 2 have GSD at 250 m, bands 3-7 have GSD at 500 m, and bands 8-36 have GSD at 1,000 m. Zheng et al. (2007a) used Landsat 7 ETM+ data and field observations to develop an empirical model. After

calibration with different sensors, MODIS data were used for model applications at a regional scale. Using a similar approach, Muukkonen and Heiskanen (2007) used ASTER (15×15 m) data to develop regression models with stand forest inventory data volume. MODIS bands 1 and 2 (250×250 m) data were used to estimate stand volume.

### *Interpretation of optical remote sensing data*

Specific interpretation procedures have been developed to extract information from images. Generally, the procedures are divided into two classes: the traditional approach using parametric methods such as regression models (Holmgren et al., 1997; Steininger, 2000), and nonparametric methods such as the k-nearest-neighbor method (k-NN) (Fazakas et al., 1999; Reese et al., 2002) (Table 2).

Since coarse spatial resolution data are difficult to couple with forest inventory measurements, researchers usually use fine or medium spatial scale resolution data to link forest inventory data to coarse spatial resolution regional data (Muukkonen and Heiskanen, 2007).

Regression models differ in variables and equations. Spectral signatures, image textures, and vegetation indexes are among the variables derived from imagery. For example, Lu and Batistella (2005) found that in the Amazon, successional forest is more likely to correlate with a spectral signature, and mature forest is more likely to correlate with texture. Zheng et al. (2007b) showed that leaf area index (LAI), and the normalized difference vegetation index (NDVI) are significant predictors for Chinese fir aboveground biomass, while LAI and stand age can predict 94% of the variation of aboveground biomass.

Regression models include linear, non-linear, multi-, and neural networks. Neural networks in forestry mainly deal with incomplete, disturbed, and noisy datasets (Hanewinkel, 2005). The neural network model was used by Steininger (2000) to develop predictive models of biomass (for example, see Figure 4). Foody et al. (2003) used multiple regression and neural networks to estimate tropical forest biomass and observed a significant relationship between predicted biomass and that measured from the forest inventories. Other researchers either use ASTER data to estimate aboveground biomass, applying non-linear regression analysis and a neural network approach (Muukkonen and Heiskanen, 2005), or fractional textures and semivariance analysis of image fractions integrated with conventional images to establish stepwise multiple regression models to predict forest structure and health (Levesque and King, 2003).

Recently, nonparametric methods such as the k-nearest-neighbor method (k-NN) and k most similar neighbor method (k-MSN) have been used to interpret images. In these methods, the prediction is no longer dependent upon the regression of the whole sample space, but on either the weighted mean of neighbors or the distance-weighted mean of most similar neighbors. The accuracy of AGB estimation was tested using the k-MSN method and was deemed acceptable (Anttila, 2002). In Sweden, Landsat data was successfully combined with the k-NN method to estimate AGB (Fazakas et al., 1999; Reese et al., 2002).

### Active remote sensing: Radar and LiDAR

Unlike optical remote sensing methods using aerial photographs and satellite images that capture the reflectance of solar radiation, Radar and LiDAR systems use their own electromagnetic radiation source independent of solar radiation. Moreover, the microwave portion of the radar wavelength can penetrate precipitation and cloud cover, and avoid shade/shadow effects from the sun (Ranson and Sun, 1994; Patenaude et al., 2005). In addition LiDAR can capture detailed stand structure and height, something difficult to achieve by the optical remote sensing method (see Table 3 for examples).

**Table 3 Selected examples of biomass estimation using radar and LiDAR data**

Datasets	Study area	Techniques	References
SIR-C	South-eastern USA	Multiple regression analysis	Harrell et al., 1997
SIR-C	Siberia	Adapted theoretical regression model	Sun et al., 2002
JERS-1 SAR L band	Ta'pajos, Para' state and Manaus, Amazonas state, Brazil	Forest backscatter regression model	Luckman et al., 1998
JERS-1 SAR L-band	New South Wales, Australia	Linear regression analysis	Austin et al., 2003
Airborne laser	Costa Rica	Linear regression, canopy height models	Nelson et al., 1997
Large-footprint LiDAR	North-east Costa Rica	Multiple regression analysis	Drake et al., 2003
Small-footprint LiDAR	Piedmont physiographic province of Virginia, south-eastern USA	Measure crown diameter using LiDAR, then estimate biomass using regression analysis	Popescu et al., 2003

Source: modified from Lu (2006).

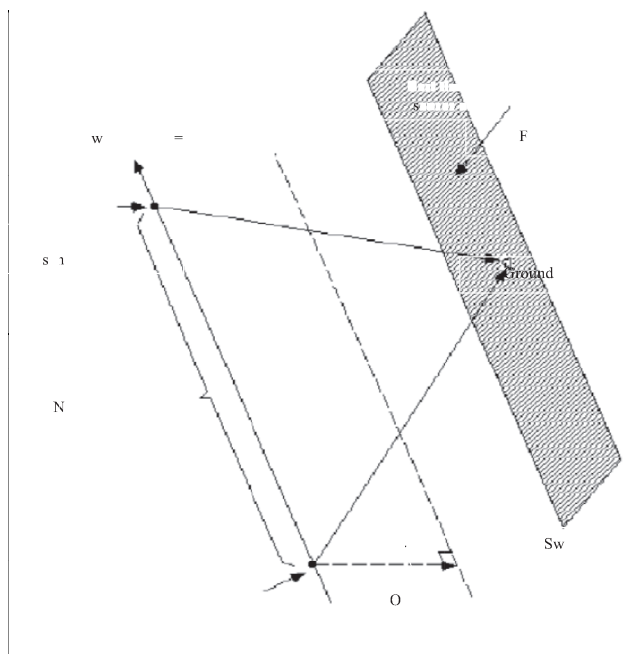
#### *Radar data*

Radio Detection and Ranging (RADAR) systems work by virtue of radiating microwave pulses to subjects and then measuring the returned echo's amplitude (backscatter amplitude) and orientation (polarization). The wavelength emitted in radar is between approximately 1 mm and 1 m. In this range, the C (3.75-7.5 cm), L (15-30 cm), and P (30-100 cm) bands are responsive, respectively, to small structural components (e.g. leaves), large components (e.g. branches), and larger components (e.g. trunks) (Patenaude et al., 2005). Unlike optical remote sensing that detects differences in reflectance of various vegetation and mineral surfaces, radar remotely detects the surface roughness, geometry, and water content of biomass.

There are two types of imaging radar, the earlier Side-Looking Airborne Radar (SLAR) and the later Synthetic Aperture Radar (SAR) (see Figure 5). SAR could be

air-, space-shuttle-, or satellite-borne and is widely used in aboveground biomass estimation. The resolution of SAR is defined in two dimensions: range and azimuth. Unlike the old SLAR radar system, whose azimuth resolution is constrained by antenna length, SAR uses signal processing to increase azimuth resolution by hundreds of times (Canada Centre for Remote Sensing, 2008). For transmitting and receiving radiation, the orientation of the electromagnetic wave (polarization) is configured as V for vertical and H for horizontal (e.g. HH is horizontally transmitted and also horizontally received waves, while VH is vertical transmitted and horizontally received radiation). Besides backscatter of amplification in different bands, polarization is also an important characteristic of predicting aboveground biomass. The horizontal and vertical distribution of the target affects the backscattered amplification of the signal (Patenaude et al., 2005).

**Figure 5** Concept of synthetic aperture




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Unlike optical remote sensing that detects differences in reflectance of various vegetation and mineral surfaces, radar remotely detects the surface roughness, geometry, and water content of biomass.

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Source: From Canada Centre for Remote Sensing, 2008. GlobeSAR-2 Radar Remote Sensing Training package. Reprinted with permission from the Government of Canada.

The interpretations of radar data mainly use regression on different variables. Properly polarized L-band SAR data are among the variables commonly used (Luckman et al., 1998; Castel et al., 2002; Sun et al., 2002).

The L-band HV (LHV) channel of the Shuttle Imaging Radar (SIR-C) data has been shown to be a strong predictor of aboveground biomass (Harrell et al., 1997; Sun et al., 2002). Likewise, the L-band HH SAR channel of the Japanese Earth Resources Satellite 1 (JERS-1) has shown a significant relationship between the backscatter coefficient of JERS-1/SAR data and the stand biomass of a pine

plantation (Castel et al., 2002). Although low correlations were found between SAR C-band backscatter and aboveground biomass, the addition of C-band HV or HH data can significantly improve estimations (Lu, 2006).

Aboveground biomass estimation by radar data can achieve good accuracy in low and medium density forests, but the relationship between radar backscatter and aboveground biomass weakens when the forest becomes too dense, reaching saturation density. Saturation density is correlated with the wavelength of band, polarization, and characteristics of the vegetation canopy and ground conditions (Lu, 2006). For example, Ranson and Sun (1994) found that L, P-band HV data appeared to saturate at 150 tons per hectare in boreal forest, while Luckman et al. (1998) found that the L-band data saturated at 60 tons per hectare in rainforest. This variability can be attributed mainly to density saturation problems rather than real differences in forest type, and emphasizes the importance of being careful when comparing and using biomass estimates derived from different band data and technologies.

#### *LiDAR data*

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Aboveground biomass estimation by radar data can achieve good accuracy in low and medium density forests, but the relationship between radar backscatter and aboveground biomass weakens when the forest becomes too dense, reaching saturation density.

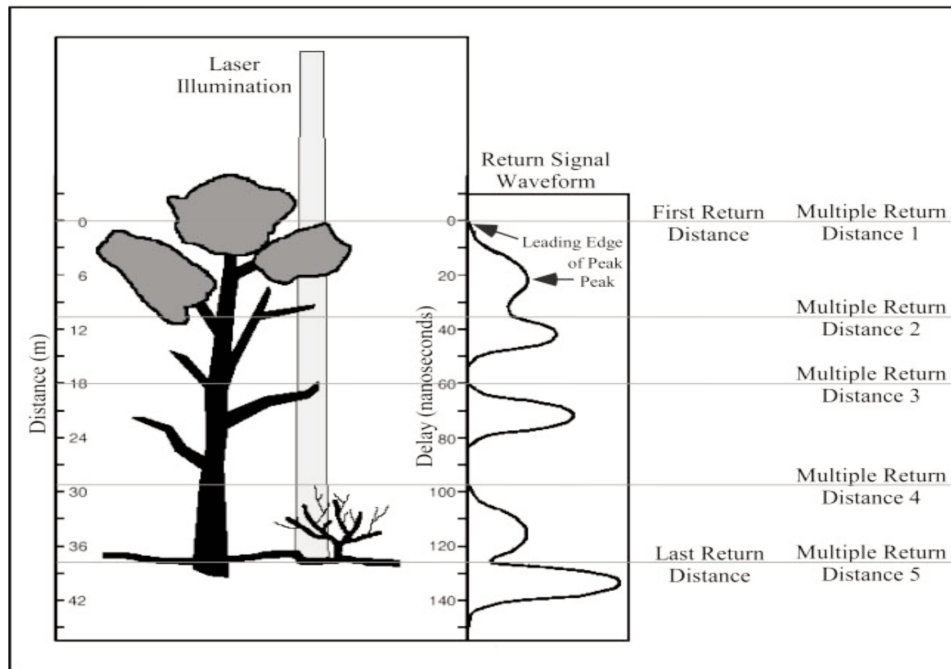
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Laser altimetry, or Light Detection and Ranging (LiDAR), is an active remote sensing method, analogous to radar, but it uses laser light instead of microwaves. The detection principle of LiDAR is similar to that of radar but is different in radiation frequency emitted. A pulse is generated with wavelengths in the visible or near infrared spectrum (900–1,064 nm), and the travel time from the sensor to the target on the ground and back is measured. Unlike optical and radar remote sensing methods, the LiDAR system provides direct information, such as the vertical structure of targets. LiDAR is therefore not actually producing images, so the data need to be converted to aboveground biomass estimations by more sophisticated models. LiDAR measurements are usually taken airborne by aircraft or helicopter (Patenaude et al., 2005).

There are two types of LiDAR systems that are distinguished by the information collected from the return signal: i) discrete-return devices (DRD); and ii) waveform recording devices (WRD). DRD can measure one (single-return systems) or a few (multiple-return systems) heights by identifying major peaks. WRD records the time-varying intensity of the returned energy from each laser pulse (Lefsky et al., 2002) (Figure 6). The DRD system has a high spatial resolution (5–90 cm) but provides limited information in stand vertical structure, while the WRD system has a low spatial resolution (10–25 m) but provides enhanced information about the vertical structure of forest.

Similarly to radar, LiDAR data are mainly used in regression models to estimate aboveground biomass. For example, studies by Nelson et al. (1997), Lefsky et al. (2002), and Drake et al. (2003) all used regression analyses to estimate aboveground biomass from mean canopy height. Wulder and Seemann (2003) tested the feasibility of using a regression model to spatially extend a LiDAR survey from a sample to a larger area with Landsat TM data. The height measured by LiDAR and correlated with Landsat TM are expected to complement the forest inventory data. At this stage, the regression models still need to be further developed (Wulder and Seemann, 2003).

Figure 6 Illustration of conceptual difference between the DRD and WRD system



Source: From Lefsky, M.A., Cohen, W.B., Parker, G.G., Harding, D.J., 2002. LiDAR Remote sensing for ecosystem studies. *BioScience* 52, 19-30. Reprinted with permission.

### Improvements for remote sensing methods

Remote sensing is a revolutionary technology for aboveground biomass estimation, with unprecedented capability of spatial, temporal, and spectral resolution and potential coverage of remote forest areas. If not restrained by cost, the data can be gathered from anywhere without political or regional restrictions, which overcomes a significant short coming of forest inventory methods for estimating aboveground biomass. Remote sensing data can also complement the conventional inventory data to increase the accuracy of models. However, to improve the utilization of remote sensing data in aboveground biomass estimation, there are several hurdles that need to be overcome.

Patenaude et al. (2005) suggest that the main potential of remote sensing is as a validation tool, rather than as a tool for producing the actual estimate of aboveground biomass, because field measurements are still needed (Fuchs et al., 2009). There are studies that have estimated aboveground biomass and compared results between inventory data and remote sensing data. In both cases MODIS and Landsat TM overestimate aboveground biomass compared with U.S Forest Inventory Analysis (FIA) (Zheng et al., 2007a; Wulder et al., 2008).

Many direct remote sensing estimations of aboveground biomass still cannot meet an acceptable accuracy without forest inventories. This could potentially be solved with better models, indexes, and instrumentation. An example of this would be

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The main potential of remote sensing is as a validation tool, rather than as a tool for producing the actual estimate of aboveground biomass, because field measurements are still needed.

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further research on the study of effects of features such as mountains, slopes, and aspects. Such features are a major source of error, and can affect vegetation reflectance, resulting in spurious relationships between aboveground biomass and reflectance. Better estimates of aboveground biomass are always made where land surfaces are flatter.

In the past, remote sensing technology has been dominated by developed nations such as the United States. However, this dependence raises the cost and risk of obtaining data worldwide and provides an over-reliance on satellites from a single country's remote sensing program. For example, reliance on the U.S. program has resulted in missed opportunities in data gathering with the failure of Landsat 6, defects in Landsat 7, the delay of LDCM, and the cancellation of vegetation canopy LiDAR. Remote sensing technology in more countries or consortiums is needed to provide the international community with more options in satellite imagery and radar/LiDAR data.

## EDDY COVARIANCE

### Basic theory and advantages

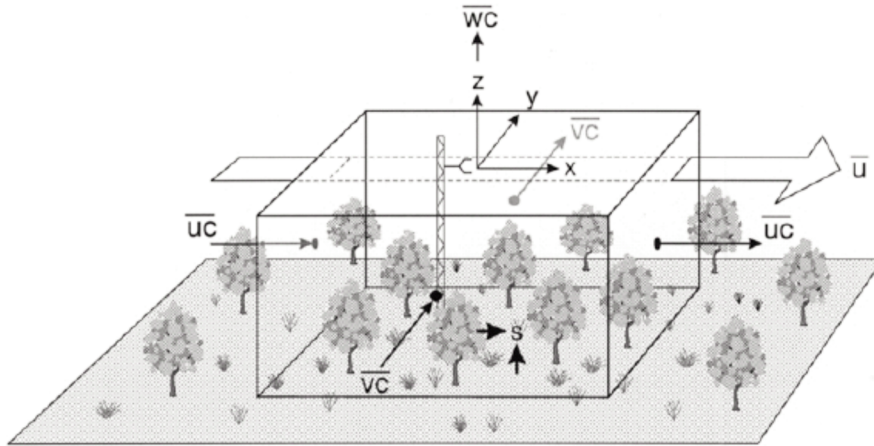
Since the late 1990s, the eddy covariance method has been developed in order to directly measure the uptake and release of CO<sub>2</sub> (CO<sub>2</sub> flux<sup>2</sup>). This method samples 3-dimensional wind speed and CO<sub>2</sub> concentration over a forest canopy at a high frequency (around 10 ~20 Hz), and determines the CO<sub>2</sub> flux by the covariance of the vertical wind velocity and CO<sub>2</sub> concentration (Moore, 1986; Gash and Culf, 1996; Bosveld and Beljaars, 2001).

The relationship between i) CO<sub>2</sub> flux and ii) the covariance of vertical wind velocity and CO<sub>2</sub> concentration is derived by putting a hypothetical control volume (box) over a homogeneous canopy (Figure 7). On the upper surface of the "box", three-dimensional wind speeds are recorded in a coordinate system that has the x axis aligned to the averaged wind direction. This assumes that one-dimensional flow (mean lateral velocity, mean vertical velocity) and stationary flow (no accumulation of CO<sub>2</sub> within the "box") is obtained over a sufficient averaging period (30 min to 1 hr). The surface exchange of CO<sub>2</sub> should then be equal to CO<sub>2</sub> exchange at the upper surface of the "box", based on the mass balance within the "box" (Finnigan et al., 2003). By measuring the vertical velocity of CO<sub>2</sub> flow at the height of the upper surface of the "box", the eddy covariance method directly measures CO<sub>2</sub> fluxes over the forest canopy (Lee, 2004, Baldocchi and Meyers, 1998).

This method is favored because of its high accuracy and appropriate spatial scale. CO<sub>2</sub> flux is usually underestimated by less than 5% during daytime and less than 12% at night. A higher accuracy can be obtained by sampling at a finer temporal and spatial resolution. For example, given normal forest canopy roughness, flat topography, and calm meteorological conditions, an anemometer positioned at 30 m with a sampling interval that is averaged every 30 to 60 minutes should provide an accurate estimate of CO<sub>2</sub> flux that covers an area from a hundred meters to several kilometers (Berger et al., 2001).

<sup>2</sup> Flux is the rate of flow of energy or particles across a given surface.

**Figure 7 Hypothetic Cartesian control volume over a homogeneous terrain.**  $V$  = vertical velocity;  $w$  = horizontal velocity;  $\vec{u}$ =velocity vector;  $c$ =a constant.



Source: From Finnigan, J.J., Clement, R., Malhi, Y., Leuning, R., Cleugh, H.A., 2003. A re-evaluation of long-term flux measurement techniques - Part I: Averaging and coordinate rotation. *Boundary-Layer Meteorology* 107: 1-48. Reprinted with permission.

Eddy covariance measurements have been continuously made at a number of sites for over ten years (Berger et al., 2001; Haszpra et al., 2005; Su et al., 2008). New observation sites, updated models, and remote sensing data enable the eddy covariance methods to continually refine estimates of CO<sub>2</sub> flux from regional to continental scales (Owen et al., 2007; Sasai et al., 2007; Yang et al., 2007; Yuan et al., 2007).

### Systematic biases

Since the eddy covariance method is derived from assumptions such as homogeneous canopy, steady environmental conditions, and stationary flow, it suffers from many systematic biases that need to be accounted for.

### Energy imbalance

For eddy covariance measurements, an imbalance exists of about 20% between turbulent energy fluxes (sensible and latent heat that is measured by the eddy covariance system) and available energy (net radiation minus stored energy that are measured separately with radiation sensors and soil heat flux plates) (Wilson et al., 2002; Han et al., 2003; Li et al., 2005).

The imbalance can be caused for three reasons: i) using 30 minutes as an averaging period in flux estimation filters out low frequency turbulence whose contribution to the flux model is missed (Foken et al., 2006); ii) flux measurements taken at different heights or across varying topographies represent CO<sub>2</sub> exchange from different source areas, with the result that the source area may not match the representative area separately measured for available energy (Schmid, 1997); and iii) the flux may not be fully detected due to advection or air drainage (Massman and Lee, 2002; Hammerle et al., 2007).

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Since the eddy covariance method is derived from assumptions such as homogeneous canopy, steady environmental conditions, and stationary flow, it suffers from many systematic biases that need to be accounted for.

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Although the CO<sub>2</sub> flux itself is not adversely affected by an energy imbalance, closing the energy budget is important for cross-site comparisons and a better understanding of underestimation and error in CO<sub>2</sub> flux measurement (Wilson et al., 2002).

### *Nighttime flux*

The boundary layer at nighttime is characterized by low wind speed, thermal stratification, and intermittent turbulence. These characteristics always cause dramatic bias in CO<sub>2</sub> flux estimations (Aubinet et al., 2005; Velasco et al., 2005; Fisher et al., 2007). Vertical and horizontal advection are not negligible, but the correction for advection is usually site-specific (Feigenwinter et al., 2008). Due to thermal stratification, CO<sub>2</sub> concentration builds up within the air layer below the measurement heights, so the storage term can also be significant. But the correction of the storage term is controversial and site-dependent, because CO<sub>2</sub> stored at night might be released in the morning when advection can be negated (Aubinet et al., 2002).

### *Topography*

Over sloping terrain, mathematical rotations of the wind coordinate system are used to meet the basic assumptions of one dimensional flow, but advection is unavoidable (Massman and Lee, 2002) and different rotation methods introduce different systematic errors to the estimation (Finnigan, 2004). Besides, CO<sub>2</sub> uptake measured at one point may be transported by drainage flows and emitted somewhere else (Sun et al., 1998).

### **Data gaps and scaling up to regions and continents**

In addition to the three systematic problems that can lead to bias in estimates, sampling intervals can be interrupted by weather (e.g. heavy rain) and other unforeseen problems such as lightning strikes. A model based on a semi-parametric relationship between net CO<sub>2</sub> flux and environmental conditions, such as light and temperature, can be used to supplement and interpolate between such data gaps (Stauch and Jarvis, 2006). Data gaps from eddy covariance measurement exist not only with sampling period (time) but also over area (space). A single eddy covariance measurement can only represent flux over hundreds meters. Multiple observation sites and sophisticated models are required to develop an estimation of regional and global CO<sub>2</sub> budgets.

Since 1998, FLUXNET, a global-scale network for eddy covariance flux measurements, was started to encourage collaboration among flux measurement sites around the globe (Baldocchi et al., 2001) (Figure 8). It supports calibration and comparison of flux measurements among sites and supports collection of vegetation, soil, hydrologic, and meteorological data for each site. Using this network, FLUXNET provides a comprehensive dataset for expanding and scaling up CO<sub>2</sub> flux estimations from a single site to global and regional estimates. However, although the number of FLUXNET tower sites has expanded from around 100 to over 400 in the last decade,

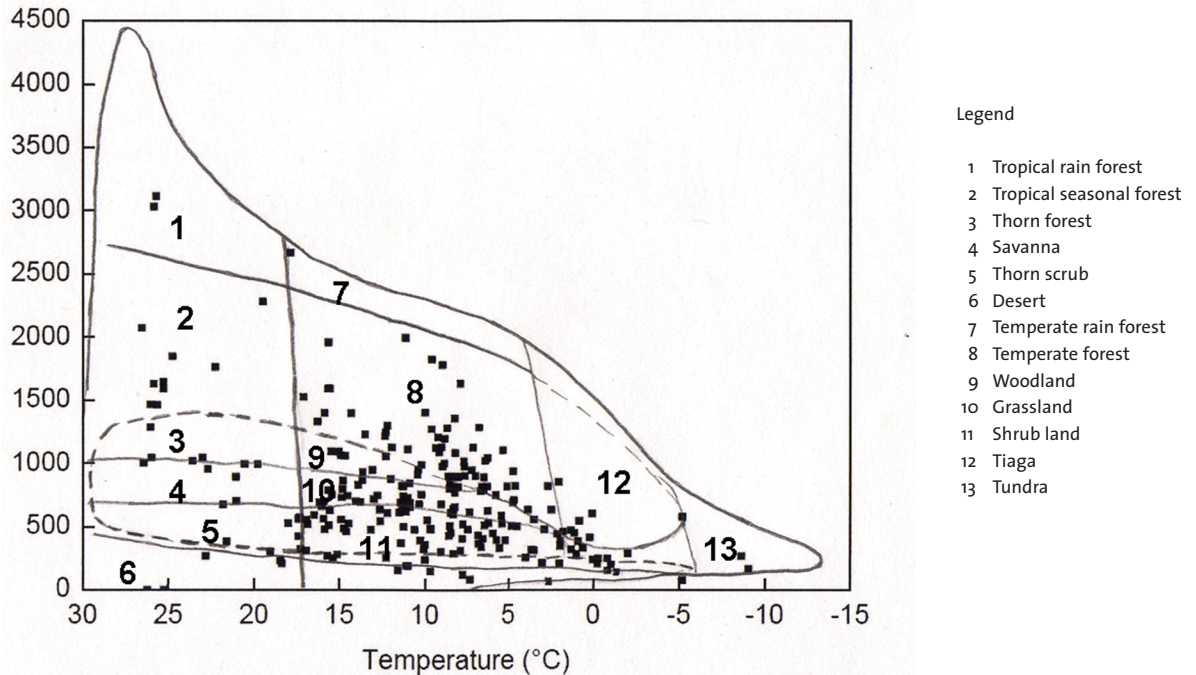
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The boundary layer at nighttime is characterized by low wind speed, thermal stratification, and intermittent turbulence. These characteristics always cause dramatic bias in CO<sub>2</sub> flux estimations

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most of the sites are located in temperate forest, grasslands, and shrubland, while measurement over some vegetation types such as tropical ever-wet and semi-evergreen rainforest, tropical dry deciduous forest, temperate rain forest, desert, urban areas, and tundra are noticeably under-represented.

Figure 8 FLUXNET sites in the climate space

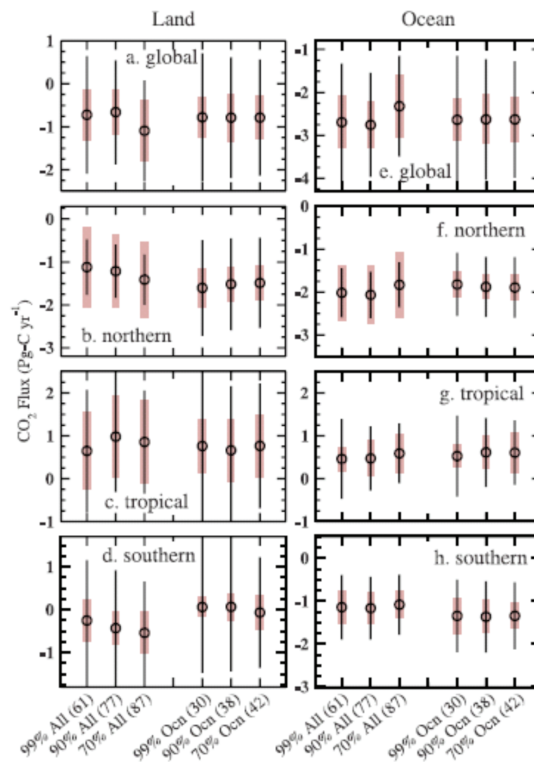


Source: Site information is from <http://daac.ornl.gov/FLUXNET/>, biome lines are drawn from Barnes et al. (1998)

Scaling models up to extend flux measurements from single sites to a larger scale involves measurements of two main processes: canopy photosynthesis and ecosystem respiration (Running et al., 1999; Soegaard et al., 2000; Wang et al., 2007b; Baldocchi, 2008). Models can be divided into two categories: i) empirical models which are based on the relationship between  $\text{CO}_2$  flux and plant eco-physiological parameters (e.g. photosynthetic light response curves); and ii) physiological growth models based on stand dynamics (Owen et al., 2007). Both categories of models can be parameterized by eddy covariance measurements, but the parameters can change considerably among different models and different ecosystems. Strong relationships between  $\text{CO}_2$  uptake and leaf area index have been utilized in the European Arctic region to calculate spatial distribution of Net Ecosystem Exchange ( $\text{CO}_2$  flux) based on Landsat TM satellite data (Soegaard et al., 2000). Still others have proposed that net ecosystem exchange may be characterized mainly by non-climatic conditions (e.g.

species, age, and site history) (Ball et al., 2007; Luyssaert et al., 2007). In a temperate moist broadleaf and coniferous forest in North Carolina, USA, parameters such as leaf nitrogen concentration and stomatal conductance were measured as inputs to a physiologically based canopy model to estimate gross primary productivity (Luo et al., 2001). Additionally, at observation sites located over heterogeneous landscapes, a footprint model has been used to determine the source area of eddy covariance measurement (Schmid, 1997; Soegaard et al., 2000; Chen et al., 2007).

**Figure 9** CO<sub>2</sub> fluxes from estimation using TransCom-3 inverse model setup and 16 global transport models. Black circles mark the average fluxes obtained from 16 models, black lines show between-model uncertainties and red thick lines show within-model uncertainties. For each panel, left part is derived from ‘all site’ data; right part is derived from ‘ocean-only’ data.



Source: From Patra, P.K., Gurney, K.R., Denning, A.S., Maksyutov, S., Nakazawa, T., Baker, D., Bousquet, P., Bruhwiler, L., Chen, Y.H., Ciais, P., Fan, S.M., Fung, I., Gloor, M., Heimann, M., Higuchi, K., John, J., Law, R.M., Maki, T., Pak, B.C., Peylin, P., Prather, M., Rayner, P.J., Sarmiento, J., Taguchi, S., Takahashi, T., Yuen, C.W., 2006. Sensitivity of inverse estimation of annual mean CO<sub>2</sub> sources and sinks to ocean-only sites versus all-sites observational networks. *Geophysical Research Letters* 33. Reprinted with permission.

In summary, eddy covariance is a promising method for both CO<sub>2</sub> flux measurements at a regional scale and CO<sub>2</sub> budget estimations at global scales. But more research needs to be conducted to close the energy budget and eliminate biases caused by night time stratification and complex topography. In addition, more sites are needed over various vegetation types that can be calibrated to other sites.

## INVERSE METHOD

Atmospheric CO<sub>2</sub> concentration can be estimated from sink and source measurements of carbon (forest inventories, flux measurements) combined with transportation models (that model gas movement) using meteorological information. It can also be measured directly. The inverse method has been developed to indirectly calculate sinks and sources of CO<sub>2</sub> from the measured concentration by using the Bayesian inversion technique (Gurney et al., 2002; Rodenbeck et al., 2003). This technique backs out carbon sources and sinks of trace gases including CO<sub>2</sub> through the use of three-dimensional transport models (Gurney et al., 2002) – hence the so-called inverse method. Transportation models and atmospheric CO<sub>2</sub> concentration data therefore determine the accuracy of the inverse method (Patra et al., 2006). Sixteen different transportation models, along with a variety of atmospheric CO<sub>2</sub> datasets, have been used to test, calibrate and estimate regional to continental scale carbon flux (Figure 9). ‘Between-model’ uncertainties are about 0.51Pg C per year, and are generally smaller than ‘within-model’ uncertainties.

The reader should be aware of the following caveats:

1. All models work better over oceans than over land.
2. Different datasets can lead to large differences in estimation. The more sites used in an inverse model, the lower the ‘within-model’ uncertainty. For example, large uncertainties in the tropical zone data reflect the few observations that are conducted there.
3. Using ‘ocean-only’ data (excluding the land and coastal measurement sites) instead of ‘all site’ data leads to better agreement between models, but the ‘within-model’ uncertainties increase.
4. Big meteorological or geological events, such as El Niño or a volcanic eruption, bias the data, leading to poor estimation.

With the development of more comprehensive datasets and improved transportation models, CarbonTracker, developed by NOAA’s Earth Systems Research Laboratory, has emerged as one of the most advanced inverse models used today (Figure 10). Over the domain covering North America and the eastern Pacific, very good agreement has been achieved between CarbonTracker predictions and real atmospheric measurements (Peters et al. 2007).

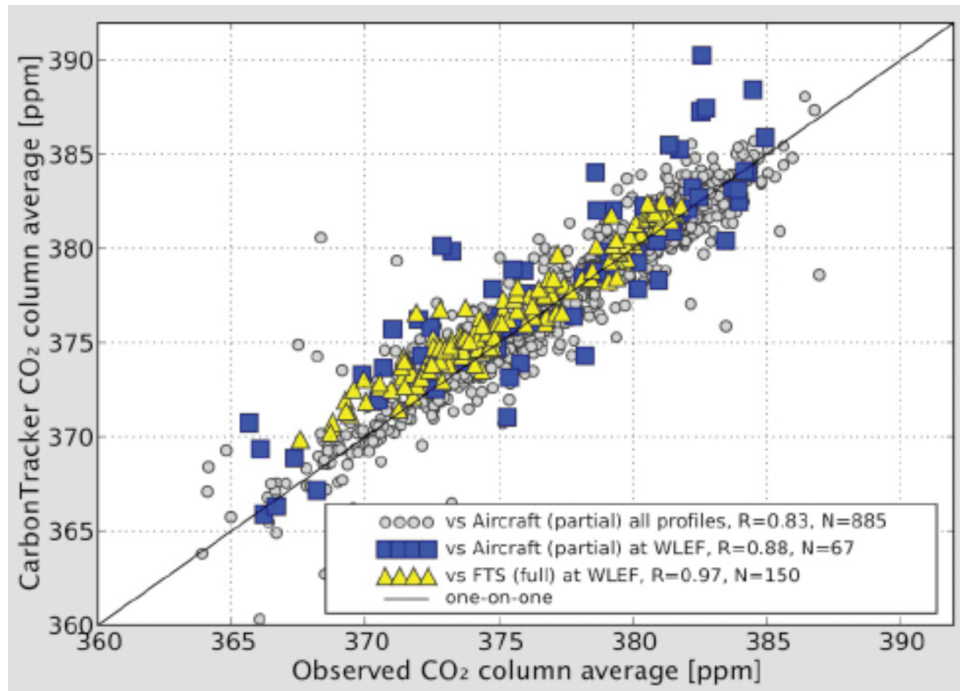
CarbonTracker is constrained by about 28,000 flask data points collected by the NOAA ESRL Cooperative Air Sampling Network and continuous CO<sub>2</sub> time series observed at several towers (Peters et al., 2007). Data processing consists of the following steps: i) develop a 3-dimensional field of atmospheric CO<sub>2</sub> mole fraction around the globe by coupling CO<sub>2</sub> surface exchange models (ocean module, fire module, fossil fuel model and biosphere model) (NOAA, 2008) with an atmospheric transport model TM5 (Peters et al., 2004; Krol et al., 2005); ii) minimize the difference between modeled and observed CO<sub>2</sub> mole fractions by adjusting linear scaling factors which control surface fluxes for large areas; and iii) build up the history of surface CO<sub>2</sub> exchange at the latitude-longitude resolution of 1°×1° (Peters et al., 2007).

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Over the domain covering North America and the eastern Pacific, very good agreement has been achieved between CarbonTracker predictions and real atmospheric measurements.

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**Figure 10 Comparison between column average CO<sub>2</sub> from observations and from CarbonTracker assimilated CO<sub>2</sub> distribution.**



Source: From Peters, W., Jacobson, A.R., Sweeney, C., Andrews, A.E., Conway, T.J., Masarie, K., Miller, J.B., Bruhwiler, L.M.P., Petron, G., Hirsch, A.I., Worthy, D.E.J., van der Werf, G.R., Randerson, J.T., Wennberg, P.O., Krol, M.C., Tans, P.P., 2007. An atmospheric perspective on North American carbon dioxide exchange: CarbonTracker. *Proceedings of the National Academy of Sciences of the United States of America* 104, 18925-18930. Reprinted with permission.

While measuring CO<sub>2</sub> concentrations, many sites also take measurements for other trace gases (e.g. methane, nitrous oxide, sulfur hexafluoride, carbon monoxide, isotopic ratios of CO<sub>2</sub> and methane). The additional measurements are not only related to climate change, but also can help in source identification of CO<sub>2</sub>. Halo-compounds (an organic compound that includes a halogen – e.g. chlorine, fluorine) and hydrocarbons (an organic compound consisting entirely of hydrogen and carbon) have recently been added to the analysis of a subset of air samples along with carbon-14, the best trace for CO<sub>2</sub> emitted through use of fossil fuels.

Although CarbonTracker is an improvement over other inverse models in many aspects, it also suffers from some problems:

- 1) The accuracy of CarbonTracker depends on the quality and number of observations available. CarbonTracker's ability to accurately quantify natural and anthropogenic emissions and uptake at regional scales is currently limited by a sparse observational network.
- 2) Predicted burned area does not match with the observed one in some regions. Methods for dealing with heteroskedastic variables through weighted least

squares or nonlinear data transformations increase the influence of low-variance observations while simultaneously decreasing the influence of high variance observations. This is undesirable for estimation (Giglio et al., 2006). Improvements need to be made in the estimation of small burned areas, although they are of less interest compared to the large burns.

- 3) In the current version of CarbonTracker, relatively small errors in fossil fuel emissions inventories are averaged out by relatively larger errors in other flux emissions (e.g. fires) (Peters et al., 2007).

In order to keep improving this tool for monitoring and predicting the global carbon cycle, all results from CarbonTracker are freely accessible, joint observations are encouraged, and models are updated every year. In addition to the simulated 3-dimensional field of atmospheric CO<sub>2</sub>, direct measurement of the 3-dimensional field from satellites is now available (Rayner and O'Brien, 2001). The satellite sensors are the Atmospheric Infrared Sounder (AIRS) and the Scanning Imaging Absorption Spectrometer for Atmospheric Cartography (SCIAMACHY) (Buchwitz et al., 2007). In 2008, two dedicated missions called the Orbiting Carbon Observatory (OCO, National Aeronautics and Space Administration) and GoSat (Japanese Space Agency) were launched to quantify CO<sub>2</sub> (Peters et al., 2004). More advanced measurements and more data will improve the performance of CarbonTracker dramatically.

## CONCLUSIONS AND RECOMMENDATIONS

The four categories of methods reviewed in this chapter are based on biomass measurement data, remote sensing data, CO<sub>2</sub> flux data (from eddy covariance) and CO<sub>2</sub> concentration data. They all exhibit their own advantages and disadvantages in estimating CO<sub>2</sub> flux and complement each other in different ways (Table 4; Figure 11).

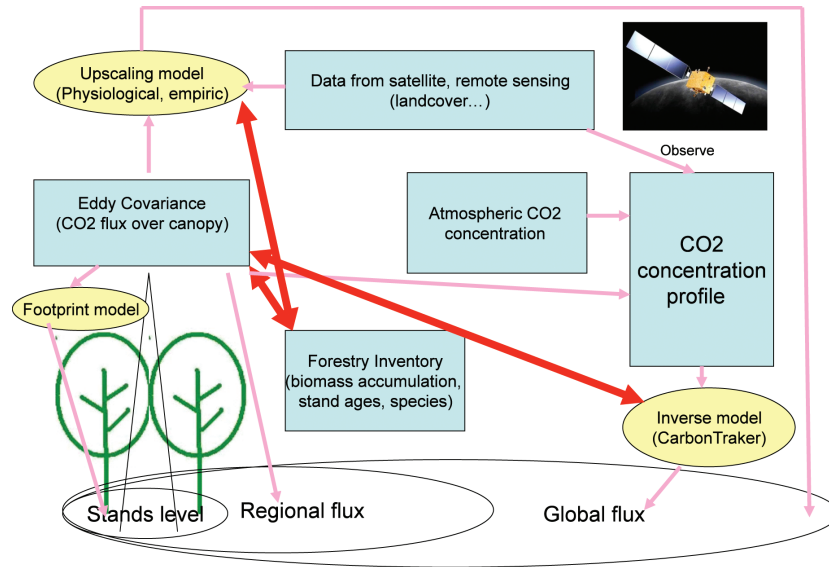
**Table 4 Summary of different methods for estimating carbon budgets**

Methods	Temporal Scale	Spatial Scale	Data Availability	Uncertainty	Target
Forest Inventory	Annual and decades	Regional	Historical data worldwide	1% for growing stock volume, 2 to 3% for net volume growth and removal, and almost 40% for change in growing stock volume.	Carbon stock in the forest
Remote sensing	Daily to annual	Regional and Global	Start from the end of 1970s	The RMSE for an aggregation area of 510 ha of forest land was 8.7% for AGB and 4.6% for wood volume.	Carbon stock in the forest
Eddy covariance	Hours to years	Over the course of a year or more	Start from end of 1960s; over 400 sites worldwide	±50gCm-2yr-1 (ideal site)	Net CO <sub>2</sub> exchange across the canopy-atmosphere interface
Inverse Method (Carbon Tracker)	Weekly	Global, at 1°×1° resolution	2000-2006	-0.65PgC/yr (for North American terrestrial biosphere)	Access net CO <sub>2</sub> exchange between the terrestrial biosphere and the atmosphere

Source: Compiled from Brown, 2002; Patenaude et al., 2005; Lu, 2006; Baldocchi, 2008; Giglio et al., 2006 and Peters et al., 2007



**Figure 11** Sketch of current observations and methods system for carbon budget (red arrow means inter comparison, pink arrow stand for data flow)



*Inventory methods* quantify biomass accumulation within forests, and are characterized by their long history and adequate data coverage (particularly in developed nations). However, they have low time resolution (years) and variable standards of measurement.

*Remote sensing methods* are most reliable if remote sensing information is jointly used with forest carbon inventories and ecosystem models. However, incomplete information limited by remote sensing techniques and uncertainties in the models require further development.

*The eddy covariance method* is advanced in its high accuracy and fine temporal resolution (hours), and is a good method for direct measurement of CO<sub>2</sub> flux at the ecosystem scale. However, it is restricted in use by its systematic biases and limited number of observation sites.

*Inverse methods* are used at continental to global scales. They retrieve the strength of both anthropogenic and non-anthropogenic sources and sinks from atmospheric CO<sub>2</sub> concentration data and transportation models. CarbonTracker is one such inverse model. The data assimilation models in these inverse methods are being improved for higher accuracy and finer spatial resolution.

No single method can meet the accuracy and resolution requirements of all users. A country, user or site will make a choice of method based on the specifics of the circumstance. To accelerate improvements, the user is encouraged to undertake data comparison, collaboration, and assimilation among different methods (Heinsch et al., 2006; Gough et al., 2008). Such improvements should build on a careful synchrony among methods. For example, CO<sub>2</sub> budget estimations from forest inventory are based on biomass accumulation, while CO<sub>2</sub> flux measurements reflect photosynthesis and

respiration – usually a one-year time lag will be found between these two results. In addition, a finer and more comprehensive observation network of CO<sub>2</sub> concentration is required.

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## Chapter 8

# The Role of Forests in Global Carbon Budgeting

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### **EXECUTIVE SUMMARY**

While forests have the capacity to sequester significant amounts of carbon, the natural and anthropogenic processes driving carbon fluxes in forests are complex and difficult to measure. However, since land use change is estimated to be the second largest source of carbon emissions to the atmosphere after the burning of fossil fuels, understanding and quantifying forest carbon sinks and sources is an important part of global carbon budgeting and climate change policy design. Although carbon emissions from land use change have remained fairly steady over the last few decades, there have been significant regional variations within this trend. Specifically, deforestation rates in the tropics, particularly in Asia, have grown significantly. In contrast, forests outside the tropics have been sequestering incremental carbon due to CO<sub>2</sub> fertilization and due to forest regrowth on lands that had been cleared for agriculture prior to industrialization.

There are several methods used to measure forest carbon fluxes; these are broadly characterized as top down or bottom up approaches. Top down approaches use atmospheric concentrations of CO<sub>2</sub> as a basis for carbon budgeting. These methods estimate global carbon pools by measuring changes in atmospheric carbon or by using atmospheric transport models to determine regional carbon fluxes across space and time. They can be useful in partitioning global carbon into oceanic and terrestrial biomes. Bottom up approaches, on the other hand, are based on forest inventories and land use change. Forest inventory models require accurate estimates of forest cover and appropriate biomass conversion factors which can be difficult due to lack of comprehensive underlying data and local variations in forest biomass concentrations. Bottom up “bookkeeping” methods, which are based on measurements of land use

change, are able to pinpoint the effects of human activity on forest carbon fluxes although they are constrained by a lack of accounting for natural disturbance.

Land use change is widely considered the most difficult component to quantify in the global carbon budget. The underlying data is often incomplete and may not be comparable across countries or regions due to different definitions of forest cover and land uses. Deforestation rates in the tropics are particularly difficult to determine due to these factors as well as differences in the way land degradation, such as selective logging and fuelwood removals, are accounted for in national statistics.

There are several knowledge and measurement gaps in forest carbon budgeting:

- Knowledge of the amount of carbon stored within each pool and across forest types is limited. Even estimates using broad categories such as carbon in vegetation versus soils vary widely due to a lack of data or assumptions about where carbon is stored within the forest and at what rate carbon is sequestered or released. Use of timber industry data such as wood volume may not be appropriate for determining the net ecosystem production of a forested area.
- Estimates of forest cover and growing stock are often based on an inadequate number of field measurement plots, particularly in the tropics. Estimation errors are further magnified by a high degree of heterogeneity in many tropical forests and by the non-normal distribution of carbon pools and fluxes.
- Carbon flux estimates from biological processes in one forest type are often applied across forest types due to a lack of alternative data despite the fact that biological processes may differ by forest type.
- Historical carbon stocks are sometimes inferred by extrapolating backwards using current data. If historic carbon inventories are inaccurate, models predicting future carbon fluxes may result in significant errors.
- Natural and anthropogenic disturbances have different impacts on forest carbon cycling over space and time. Carbon flux estimates that do not distinguish by type of disturbance may generate erroneous estimates of disturbance and post-disturbance related carbon fluxes.

While significant challenges remain in quantifying forest carbon pools and fluxes, these challenges become even more difficult under climate change:

- Climate change is likely to generate both positive and negative feedbacks in forest carbon cycling. Positive feedbacks may include increased fire and tree mortality from drought stress, insect outbreaks and disease. Negative feedbacks may include increased productivity from CO<sub>2</sub> enrichment. While the net result from positive and negative climate feedbacks is generally thought to be higher net carbon emissions from forests, the timing and extent of these net emissions are difficult to determine.

- While forests may exhibit greater rates of photosynthesis due to higher levels of CO<sub>2</sub> in the atmosphere, at some point this increased productivity will be inhibited by nutrient limitation. At which point this occurs is likely to differ by region and forest type.
- Temperature increases are likely to have multiple compounding and offsetting impacts which make it difficult to quantify the net impact on carbon cycling. While longer growing seasons may increase carbon uptake in forests, warmer temperatures may lead to increased drought which could offset any increased sequestration from a longer growing season although evidence of this is equivocal. In addition, warmer temperatures may lead to increased carbon and methane emissions from thawing peatlands.
- The frequency and severity of disturbances are likely to increase. Estimating forest carbon fluxes following disturbance will be difficult if changes in temperature, precipitation, and species composition lead to forest recovery patterns that deviate from historical patterns.

Given the uncertainties in forest carbon budgeting, there are several recommendations for policymakers seeking to use carbon budgeting to design forest policy. First, models should be selected based on the carbon pools and processes under consideration. Carbon policies should be tested using multiple methodologies to avoid unintended consequences. Second, greater numbers of permanent, long term research plots should be created to improve knowledge of carbon processes and to better estimate carbon fluxes across spatial and temporal gradients. Third, countries should be required to adhere to globally accepted methodologies for determining forest cover, land use, and biomass conversion factors. Fourth, regionally specific carbon data should not be extrapolated to other regions and forest types. Finally, policymakers should consider the immediate impacts of policies on forest carbon fluxes as well as the longer term impacts to ensure long term carbon management goals are met.

## INTRODUCTION

Quantifying carbon sources and sinks is a particular challenge in forested ecosystems due to the roles played by biogeochemistry, climate, disturbance and land use, as well as the spatial and temporal heterogeneity of carbon sequestration across regions and forest types. Nevertheless, as emissions from land use change (largely deforestation) are a significant percentage of the overall global carbon budget, the role of forests continues to be a key component of global carbon policy design.

Forests can act as a sink or a source of carbon under different conditions and across temporal and spatial gradients. Understanding the role of forests in global carbon budgets requires quantifying several components of the carbon cycle, including how much carbon is stored in the world's forests (carbon pools), gains and losses of carbon in forests due to natural and anthropogenic processes (carbon

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Understanding the role of forests in global carbon budgets requires quantifying several components of the carbon cycle, including how much carbon is stored in the world's forests (carbon pools), gains and losses of carbon in forests due to natural and anthropogenic processes (carbon fluxes), exchanges between the terrestrial carbon and other sinks and sources, and the ways in which such processes may be altered by climate change.

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fluxes), exchanges between the terrestrial carbon and other sinks and sources, and the ways in which such processes may be altered by climate change.

This chapter will review the current research in forest carbon budgeting. It will consider the tools used to quantify forest carbon pools and fluxes and their relationship to the global carbon budget. It will demonstrate the complexity of terrestrial carbon sequestration and its interdependence with other components of the carbon cycle by highlighting gaps in knowledge, measurement tools, and models. Finally, it will conclude with some recommendations for future research to better understand forests and their role in global carbon budgeting.

### THE GLOBAL CARBON BUDGET

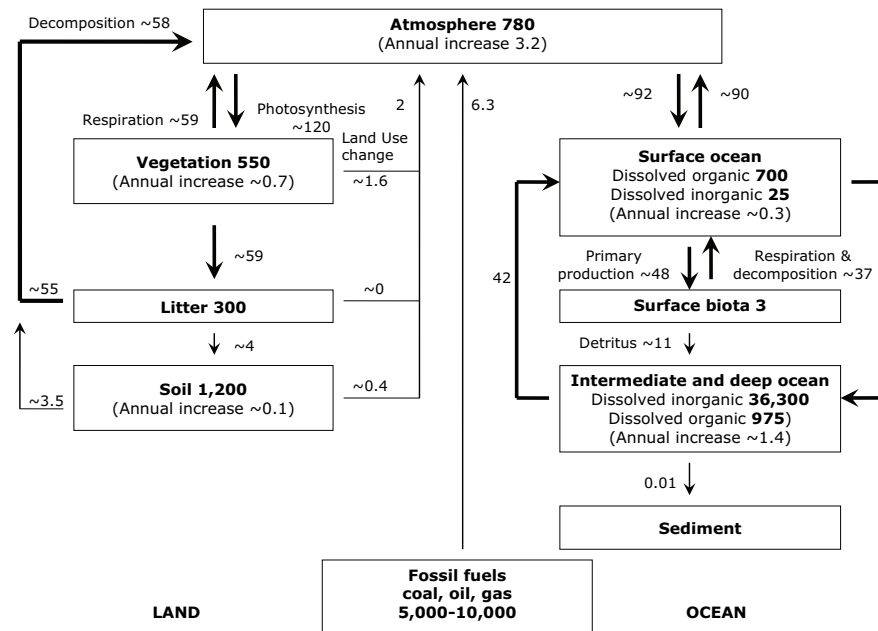
The world's carbon is stored in four primary pools. These include oceans (38,000 PgC), fossil fuels (5,000-10,000 PgC), terrestrial ecosystems (1650 - 4000 PgC), and the atmosphere (805 PgC) (Houghton, 2007). Carbon flows between these pools through natural processes (such as photosynthesis, respiration, and decomposition) and anthropogenic processes (such as burning of fossil fuel and human-induced land use change) (Figure 1).

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Figure 1 The global carbon cycle (1990s)



Source: Houghton, R.A., 2007. Balancing the global carbon budget, *Ann. Rev. Earth Planet Sci.* 35, 313-347. Reprinted with permission.



It is well documented that the largest source of carbon emissions to the atmosphere stems from the burning of fossil fuels (Table 1). Due to economic growth, population growth, and an increase in the energy intensity of gross domestic product (industrialization), fossil fuel emissions have been on a rising trend, particularly since 2000 (Raupach et al., 2007). Some portion of these emissions is removed from the atmosphere by ocean and terrestrial carbon sequestration processes. However, in the last several years, measurements indicate that carbon emissions have grown faster than land and ocean sinks, and that the efficiency of the ocean carbon sink is declining (Fung et al., 2005). Although oceans will increase carbon uptake with rises in atmospheric CO<sub>2</sub>, their rate of uptake is limited by higher temperatures (coinciding with increased atmospheric CO<sub>2</sub>), reduced vertical mixing, and reduced buffering capacity (Prentice, 2001; Sarmiento et al., 1998).

Canadell et al. (2007) have cited an increase in the airborne fraction (the ratio of the annual increase in atmospheric CO<sub>2</sub> to CO<sub>2</sub> emissions from anthropogenic sources) as evidence that global ocean sinks are weakening or, in some cases, becoming sources (Table 1). This claim is supported by evidence that suggests that climate change-related temperature and precipitation increases, along with changes in wind patterns, have reduced the oceans' efficiency as a carbon sink (Anderson et al., 2009; Caldeira and Duffy, 2000). In the Southern Ocean, strong, westerly winds induce upwelling of deep, carbon rich waters and expose this carbon to the atmosphere (Anderson et al., 2009; Metz et al., 2009). LeQuere et al. (2007) estimate that between 1982 and 2004 CO<sub>2</sub> sequestration in the Southern Ocean weakened by 0.08 PgC per year. They attribute this phenomenon to stratospheric ozone depletion, intensification of Southern Ocean winds, and changes in ocean surface temperature gradients, all largely driven by human-induced climate changes (LeQuere et al., 2007, Thompson and Solomon, 2002).

**Table 1 Trends in the global carbon budget over time**

Global Carbon Budget	Mean			
	1959-2006	1970-1999	1990-1999	2000-2006
<b>Economy (kgC/US\$)</b>				
Carbon Intensity	0.29*	0.30	0.26	0.24
<b>Sources (PgC/yr)</b>				
Fossil Fuel	5.30	5.60	6.50	7.60
Land Use Change	1.50	1.50	1.60	1.50
Total	6.70	7.00	8.00	9.10
<b>Sinks (PgC/yr)</b>				
Atmosphere	2.90	3.10	3.20	4.10
Ocean	1.90	2.00	2.20	2.20
Land	1.90	2.00	2.70	2.80
<b>Airborne Fraction</b>	0.43	0.44	0.39	0.45

\* data from 1970

Source: Canadell, J.G., LeQuere, C., Rapach, M.R., Field, C.B., Buitenhuis, E.T., Ciais, P., Conway, T.J., Gillett, N.P., Houghton, R.A., Marland, G., 2007. Contributions to accelerating atmospheric CO<sub>2</sub> growth from economic activity, carbon intensity and efficiency of natural sinks. PNAS, 104:47, 18866-18870. Reprinted with permission.

The second largest source of emissions results from land use change. While aggregate emissions from land use change have been fairly constant over the last several decades, the aggregated figures have masked significant regional and temporal variability in emissions trends (Table 2). From 1959-1980, approximately two-thirds of land use-related emissions originated from the tropics, resulting from deforestation in the Americas, Asia, and Africa (Houghton, 2003b). This trend began to change in the 1980s and 1990s when net land use change emissions outside the tropics fell to zero as forest regrowth sequestered sufficient amounts of carbon to offset emissions from disturbance and management practices (Goodale et al., 2002; Myneni et al., 2001). In the 1990s, tropical deforestation-related emissions have grown in Asia, fallen modestly in the Americas, and have risen in Africa (DeFries et al., 2002).

**Table 2 Carbon fluxes from land use change**

<i>Region</i>	<i>Total (PgC) 1850-2000</i>	<i>Annual Flux (PgC/yr)</i>	
		<i>1980-1989</i>	<i>1990-1999</i>
<b>Tropical Asia</b>	48	0.88 ± 0.5	1.09 ± 0.5
<b>Tropical America</b>	37	0.77 ± 0.3	0.75 ± 0.3
<b>Tropical Africa</b>	13	0.28 ± 0.2	0.35 ± 0.2
<b>Total Tropics</b>	98	1.93 ± 0.6	2.20 ± 0.6
<b>Canada</b>	5	0.03 ± 0.2	0.03 ± 0.2
<b>US</b>	7	(0.12) ± 0.2	(0.11) ± 0.2
<b>Europe</b>	5	(0.02) ± 0.2	(0.02) ± 0.2
<b>Russia</b>	11	0.03 ± 0.2	0.02 ± 0.2
<b>China</b>	23	0.11 ± 0.2	0.03 ± 0.2
<b>Pacific Developed</b>	4	0.01 ± 0.2	0.00 ± 0.2
<b>North Africa/Mid East</b>	3	0.02 ± 0.2	0.02 ± 0.2
<b>Total Ex Tropics</b>	58	0.06 ± 0.5	(-0.02) ± 0.5
<b>Global Total</b>	156	1.99 ± 0.8	2.18 ± 0.8

While aggregate emissions from land use change have been fairly constant over the last several decades, the aggregated figures have masked significant regional and temporal variability in emissions trends

Source: Houghton, R.A., 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850-2000. *Tellus* 55B, 378-390. Reprinted with permission.

## MODELING GLOBAL CARBON BUDGETS

### Top down approaches

Although scientists agree on the broad categories of emissions and sources, there are different methods used to quantify carbon sinks and sources, which often produce a wide variety of results. Each method accounts for carbon processes in different ways and demonstrates that there remain significant gaps in measurements and knowledge of the terrestrial carbon cycle (see Chapter 7, this volume for more details on measurement methodologies). Taken together, however, carbon budget models can be complementary and help to provide a more comprehensive picture of the global carbon budget.

Global carbon models generally fall into two methodological categories: top down and bottom up approaches (Schimel, 2007; Peylin et al., 2005). Top down approaches

start with atmospheric concentrations of CO<sub>2</sub> to build a full accounting of the global carbon budget. Assessment Reports by the Intergovernmental Panel on Climate Change (IPCC) use top down approaches to measure global carbon sinks and sources (Nabuurs et al., 2007). There are two main types of top down models. The first, used by the IPCC, seeks to partition carbon sinks on land and in oceans by measuring changes in atmospheric concentrations of O<sub>2</sub>/N<sub>2</sub> alongside measurements of <sup>13</sup>C/<sup>12</sup>C ratios (Keeling et al., 1996). While ocean uptake of CO<sub>2</sub> does not meaningfully impact O<sub>2</sub>/N<sub>2</sub> ratios in the atmosphere, in the terrestrial biome the burning of fossil fuels decreases O<sub>2</sub>/N<sub>2</sub>, although some of this is offset by terrestrial releases of O<sub>2</sub> from plant growth (Battle et al., 2000). Similarly, the isotopic composition of carbon (also known as isotopic fractionation) does not change due to fluxes between the air and oceans (Ciais et al., 1995). In contrast, photosynthesis discriminates against <sup>13</sup>C, although measurements must consider the spatial distribution of C<sub>3</sub> and C<sub>4</sub> plants since they discriminate against <sup>13</sup>C differently and will therefore influence <sup>13</sup>C/<sup>12</sup>C ratios (Manning and Keeling, 2006). Despite these nuances, analyzing changes in these ratios is a generally accepted methodology to partition land and ocean carbon sinks.

Critics of this model have pointed out that it must be adequately adjusted to account for oceanic outgassing of O<sub>2</sub>, otherwise the terrestrial carbon sink will be overstated (Plattner et al., 2002). Inadequate accounting for O<sub>2</sub> outgassing in the third IPCC report is thought to explain in part why terrestrial carbon sinks in the report show a large increase from the 1980s to the 1990s (Manning and Keeling, 2006; Keeling et al., 1996). Researchers such as Plattner et al. argue that proper accounting for oceanic outgassing would reduce the IPCC terrestrial carbon sink in the 1990s from 1.4 PgC/yr to 0.7 PgC/yr, which is more consistent with the net carbon terrestrial sink of 0.4 PgC/yr recorded in the 1980s (Plattner et al., 2002). Other studies suggest the adjustment is closer to 0.2-0.3 PgC/yr, which reduces the IPCC estimate of the terrestrial carbon sink to 1.2 PgC/yr during the decade (Manning and Keeling, 2006).

The second top down approach is called *inverse modeling*. Inverse models also examine atmospheric concentrations of CO<sub>2</sub>. However, they measure regional distributions of carbon concentrations across space and time and use atmospheric transport modeling to estimate global sources and sinks. Inverse models are heavily influenced by the type of atmospheric transport model used, assumptions about prior regional fluxes, time resolution (annual versus monthly data), spatial resolution (number of source regions), and an ability to reconcile seasonal variations which may impact measurement of carbon fluxes in northern versus southern hemispheres (Peylin et al., 2002; Schimel et al., 2001).

Inverse modeling has been used to try and pinpoint the “missing sink,” also called the residual terrestrial sink. This residual carbon sink represents the excess carbon accumulation that has not been directly observed but is required to balance the carbon budget (Figure 2). Atmospheric modeling techniques generally attribute this missing sink to a large carbon accumulation in northern mid-latitude terrestrial ecosystems (Fan et al., 1998). This is thought to be the result of forest re-growth in areas that had previously been cleared for agriculture as well as increased productivity from CO<sub>2</sub> fertilization (Alexandrov et al., 1999; Fan et al., 1998; Caspersen et al., 2000).

More recent research has suggested that inverse models have not adequately adjusted for seasonal variations in CO<sub>2</sub> fluxes in the Northern Hemisphere, leading to biases in annual-mean fluxes which have overstated the terrestrial carbon sink in northern latitudes (Stephens et al., 2007). Thus, while land use-related emissions may be high in the tropics, there exist strong carbon sinks in undisturbed areas of the tropics that may be mistakenly attributed to temperate regions. The contribution of the “missing” carbon sink over time is seen in Figure 3.

Figure 2 The global carbon cycle equation

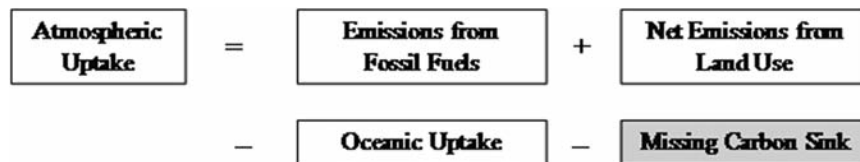
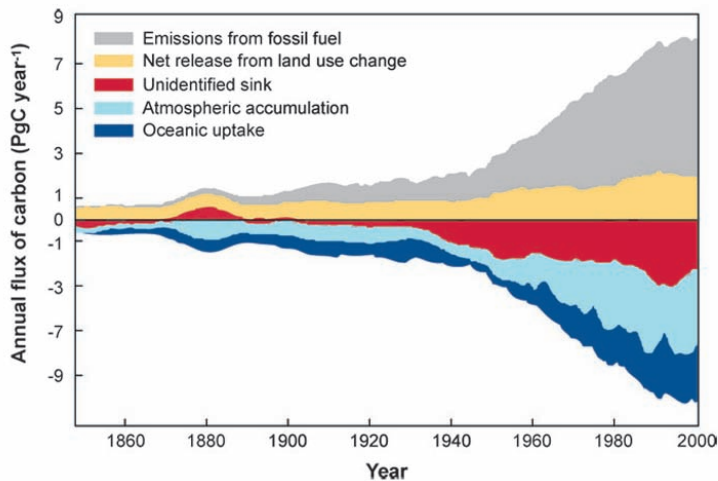


Figure 3 The missing sink in the global carbon budget



Source: Houghton, R.A., 2007. Balancing the global carbon budget, *Ann. Rev. Earth Planet Sci.* 35, 313-347. Reprinted with permission.

While land use-related emissions may be high in the tropics, there exist strong carbon sinks in undisturbed areas of the tropics that may be mistakenly attributed to temperate regions.

Some policymakers are fearful of using inverse modeling to set global greenhouse gas emissions reduction targets. They argue that countries in the northern hemisphere would be able to claim that emissions reduction efforts in their region are less critical due to the strong existing sink they provide in the global carbon budget (Kurz et al., 2008a). Others have suggested that because inverse models measure only net fluxes, they mask impacts from gross emissions sources that make it difficult to model future carbon fluxes from land clearing and complicate efforts to create effective policy governing land use change.

Since inverse models produce net results of carbon sinks and sources, they do not tend to isolate components of the net figures. In the tropics, therefore, gross sources

from deforestation are often not clearly delineated (DeFries et al., 2002). In addition, carbon in terrestrial materials is transported by riverine systems into oceans, where it is released by air-sea gas exchange. Inverse models, however, may erroneously record the carbon transported by riverine systems as a “sink” in the terrestrial balance while showing a “source” from the oceanic side, thereby leading to an overestimation of the net terrestrial sink (Aumont et al., 2001; Pacala et al., 2001). This highlights the challenges of inferring carbon storage from flux measurements. Fluxes do not necessarily equate to carbon storage; thus, it is important to adequately trace fluxes in order to determine whether measured fluxes on land or in the ocean are indeed matched with changes in underlying carbon pools in those areas (Houghton, 2007).

### Bottom up approaches

Bottom up approaches, on the other hand, are based on forest inventories and land use change. In inventory modeling, data on forest area, timber stocks, and forest growth are converted to biomass estimates to determine the carbon density of the vegetation, and then aggregated to form the forest carbon budget (Dixon et al., 1994). The robustness of forest inventory methods is a function of accurate estimates of forest cover and appropriate biomass conversion factors. Variability in forest cover and biomass estimates, however, can vary widely, as seen in Table 3.

As an example, the Food and Agriculture Organization of the United Nations (FAO) publishes Forest Resource Assessments (FRAs) using inventory methods. In the 2000 assessment, more than 650 definitions of forest were assembled from 132 developing countries, using 110 independent surveys. Using these data, forest area was determined by using a 10% canopy cover threshold (FAO, 2000). Estimation errors can result due to inconsistent definitions of forest cover, incomplete or incorrect data from national sources, and the accounting (or omission) of certain types of land, such as recently disturbed forested area which have temporarily fallen below 10%, or areas of sparse tree cover (Kauppi, 2003; Brown, 2002).

Challenges to inventory methods generally point to the fact that forest inventory data is incomplete, particularly in the tropics, and that the estimates based on data limited in scope (such as wood volume) can lead to significant errors. First, inventory models do not consider carbon on lands that have been heavily impacted by management (Houghton, 2003b). Second, there is a lack of adequate accounting for belowground biomass, soils, litter, and forest products which is magnified by the temporal and spatial heterogeneity of these forest carbon components (House et al., 2003, Brown, 2002). In many cases, non-timber forest carbon pools, such as soils, are inferred through model estimates rather than measured directly (Houghton 2003a). There is also inadequate accounting of fuelwood removals, slash burns, and understory vegetation (Turner et al., 1995). Many have argued the need for a global network of permanent plots as well as improved remote sensing technology to better understand carbon fluxes from the different components of forested ecosystems (Dixon et al., 1994) (see Chapter 7, this volume, for a detailed discussion of remote sensing methods). On a broader scale, researchers have questioned the efficacy of inventory data due to methodological differences in country level accounting (Dong

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The robustness of forest inventory methods is a function of accurate estimates of forest cover and appropriate biomass conversion factors.

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et al., 2003; Myneni et al., 2001), although FRA assessments seek to conform original country data, based on national definitions and sampling techniques, to international, comparable statistics (FAO, 2000). In terms of the model itself, critics point out that conversion factors used to translate wood data into carbon may or may not be accurate, depending on whether they take into consideration such factors as the age class distribution of the stand, species composition, and whether it is a natural or managed stand (Goodale et al., 2002; Alexandrov et al., 1999).

**Table 3** Estimates of forest area and biomass

	FRA 2000 and Fung et al (2001)	Houghton (1996, 1999)
<b>Tropical</b>		
Forest area (10 <sup>6</sup> ha)	1871	2167
Woody biomass (PgC)	164 <sup>1</sup>	288
Biomass per area (kgC/m)	8.8	13.3
<b>Non-Tropical</b>		
Forest area (10 <sup>6</sup> ha)	1998	2659
Woody biomass (PgC)	93 <sup>2</sup>	223
Biomass per area (kgC/m)	4.7	8.4
<b>Total</b>		
Forest area (10 <sup>6</sup> ha)	3869	4827
Woody biomass (PgC)	257	510
Biomass per area (kgC/m)	6.6	10.6

1) Tropical pool is estimated as the total (257) minus non-tropical (93) = 164 PgC.

All forests of South America and Africa are included in "tropical forest".

2) 88PgC excluding China and 475 PgC estimated for China (Fang et al, 2001).

All forests of China, Australia and the US are included in "non-tropical".

Source: Data from Kauppi, P., 2003. New low estimate for carbon stock in global forest vegetation based on inventory data. *Silva Fennica* 37:4, 451-458.

Other bottom up models include "bookkeeping" and "process based" models (Ramankutty et al., 2007; Houghton, 2005a). Bookkeeping models track changes in below- and above-ground carbon stocks through changes in land use. Such models have been viewed as complementary to top down models because, unlike models based on atmospheric data, bookkeeping models are able to isolate changes in carbon stocks specifically driven by human activity (such as land clearing) (Houghton 1999). Bookkeeping models measure carbon fluxes from anthropogenic land use stemming from releases at the time of clearing, but also include slower carbon releases from residual debris, as well as sequestration from regeneration and regrowth, and changes in soil carbon stocks (Achard et al., 2004). This allows researchers to effectively track the "fate of carbon" over time as a result of land use activities (Ramankutty et al., 2007). Not surprisingly, this is often seen as a valuable input to land management and carbon policymaking, given the need to manage emissions resulting from land use change.

Since bookkeeping models are constrained to land use change related carbon sources, they typically exclude emissions from natural disturbances, which can be a significant carbon source. For example, an analysis of carbon uptake by U.S. forests in the 1980s concluded that forest regrowth was the overwhelmingly dominant driver

of carbon uptake, accounting for nearly all of the net sequestration during the period (Caspersen et al., 2000). A bookkeeping analysis of the same region during the same decade estimated that forest regrowth following past harvests accounted for only 20-30% of carbon uptake in U.S. forests during the same period (Houghton et al., 1999). The remaining 70% was undetermined since it was outside the scope of land use change. Both analyses claimed that nutrient enrichment was a small driver of incremental carbon uptake. Thus, the difference between two studies must therefore be attributed (either partially or entirely) to the role of recovery from natural disturbance, which is not accounted for in bookkeeping methods.

Similarly, carbon sequestration rates measured in Canadian boreal forests recently harvested have been lower than predicted sequestration rates using land use-based models (Kurz et al., 1999). In addition, the actual rate of emissions from some forests has been greater than the predicted rate using bookkeeping methods. This has been attributed to fires and insect outbreaks which are not counted in bookkeeping methods (Houghton, 2003b, Kurz et al., 1999).

Some of the differences between top down and bottom up estimates of carbon fluxes can be seen in Table 4.

**Table 4 Annual carbon fluxes estimated by top down and bottom up models (negative values = carbon sink)**

	<b>O<sub>2</sub> and CO<sub>2</sub></b>	<b>Inverse Calculations CO<sub>2</sub>, <sup>13</sup>CO<sub>2</sub>, O<sub>2</sub></b>	<b>Forest Inventories</b>	<b>Land-use Change</b>
<b>Globe</b>	-0.7 (±0.8) <sup>1</sup>	-0.8 (±0.8) <sup>2</sup>	--	2.2 (±0.6) <sup>3</sup>
<b>Northern mid-latitudes</b>	--	-2.1 (±0.8) <sup>4</sup>	-0.6 <sup>5</sup>	-0.03 (±0.5) <sup>3</sup>
<b>Tropics</b>	--	1.5 (±1.2) <sup>6</sup>	-0.6 (±0.3) <sup>7</sup>	0.5 to 3.0 <sup>8</sup>

1) Plattner et al, 2002

2) -1.4 (±0.8) from Gurney et al 2002, reduced by 0.6 to account for river transport (Aumont et al, 2001)

3) Houghton, 2003b

4) -2.4 (Gurney et al, 2002), reduced by 0.3 to account for river transport (Aumont et al, 2001)

5) Forests only, including wood products (Goodale et al, 2002)

6) 1.2 from Gurney et al, 2002, increased by 0.3 to account for river transport (Aumont et al, 2001)

7) Undisturbed forests (Phillips et al, 1998, Baker et al, 2004)

8) Fearnside, 2000, DeFries et al, 2002, Houghton, 2003b, Achard et al, 2004

Source: Houghton, R.A., 2007. Balancing the global carbon budget, *Ann. Rev. Earth Planet Sci.* 35, 313-347. Reprinted with permission.

Comparing the results of different models can often highlight uncertainties in both the data and in the models themselves. Results from inversion models attribute a large amount of carbon uptake to northern latitudes, versus land use change-based inventory methods that do not show as large a sink. Some of this may be due to the seasonal covariance between terrestrial carbon fluxes and atmospheric transport, which can lead to overestimations of northern terrestrial carbon storage (Houghton, 2007; Gurney et al., 2004). At the same time, as mentioned above, models that are based on human-induced land use change may not account for sinks or sources from natural processes such as insect outbreaks and CO<sub>2</sub> fertilization. This does not imply

that land use models are inadequate. Rather, bottom up estimates used in conjunction with top down carbon estimates can help to paint a fuller picture of global carbon sinks and sources than would be seen using the methods in isolation.

### CHALLENGES IN ESTIMATING LAND USE CHANGE

Land use change is widely considered the most difficult component to quantify in the global carbon budget (Canadell et al., 2007). Since land use change is estimated to be the second largest source of carbon emissions to the atmosphere, however, it is critical that policymakers have sufficient tools to understand its carbon dynamics. Estimates of carbon fluxes from global trends in land use change are shown in Table 2.

Most scientists agree that land use change (deforestation) in the tropics has caused the tropical biome to become a net carbon source, while outside the tropics, changes in land use (primarily reforestation), but also changes in forest age and structure have resulted in the temperate and boreal biomes becoming a net carbon sink. However, there are significant challenges both in the data themselves and the way they are appropriated in carbon budgeting. Regional, national, and local data vary widely, particularly on a historical basis (Houghton, 2007). Data sources do not always reconcile, as in the case of China where FAO data show cropland on a rising trend while the USDA data show the opposite (Houghton, 2003b). To the extent that measurements of forest area are inferred from data taken from other terrestrial sources such as croplands, there is significant scope for estimation error. This, in turn, can potentially cause erroneous conclusions about assumed carbon sinks and sources from different forest management activities.

The ability to compare data across regions is also difficult due to differences in country-specific definitions of forest cover (Waggoner, 2009). This is one reason why researchers caution the use of Forest Resource Assessment data in measuring changes in land use (DeFries et al., 2002). Researchers also point out that FAO country level assessments are often based on local inventories which may be of insufficient sample size, outdated, and not representative of forest cover at a national scale (Houghton, 2005; Matthews, 2001).

Some have argued that remote sensing data can be used to ground truth and fill gaps in forest cover data, particularly in the tropics (Mayaux et al., 2005). Even with remote sensing data, however, it may be difficult to reconcile differences in the measured extent of forest cover. For example, estimates of forest cover in Russia based on remote sensing data are substantially lower than estimates using FAO data. This is likely due to differences in how forestland is defined by both methods as well as satellite data resolution which may not capture forests in tundra regions (Dong et al., 2003).

It is also important to consider the temporal dynamics of land use change, in terms of historical, current and future impacts on carbon fluxes. Most scientists agree that northern mid-latitude forests have shown a net carbon accumulation over the last several decades. However, there is less agreement on the processes driving this uptake.

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Comparing the results of different models can often highlight uncertainties in both the data and in the models themselves.

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Many researchers attribute this sink to secondary forest growth as a result of past land use, such as in the northeastern United States where forests have regenerated following the abandonment of agricultural lands (Barford et al., 2001; DeFries et al., 2002; Schimel et al., 2001). Since lands deforested for agriculture were initially a carbon source, today's carbon sink is an "inherited" carbon uptake linked to past carbon emissions. It is important to distinguish inherited sinks driven by past land use from sinks driven by biophysical processes such as CO<sub>2</sub> enrichment or longer growing seasons stemming from climate change (Schimel, 2007). While both potentially serve as strong drivers of carbon uptake, their long term trends may be quite different. Studies have shown that carbon sequestration rates are often higher on lands recovering from disturbance (or intensive management) versus long term accumulations on unmanaged, natural landscapes (Schimel et al., 2001). Thus, understanding the underlying mechanisms driving carbon sequestration rates is not only necessary for accurately projecting carbon uptake rates from secondary forests in temperate regions, particularly as they mature, but in projecting long term carbon fluxes in tropical regions where current areas of deforestation may become sinks if agricultural lands are converted back to forest.

In addition, emissions from land use change have their own temporal variation. In short, how land is cleared influences when and how much carbon is emitted to the atmosphere. For example, slash and burn clearing for agriculture tends to result in higher emissions in earlier years versus harvests which convert timber to long lived wood products (Ramankutty et al., 2007). Thus, despite the fact that both activities fall into the category of land use change, the carbon fluxes observed over time may be quite different. Estimating future fluxes therefore requires making assumptions about the type of land use change expected to dominate in a particular area.

Perhaps the greatest amount of debate, however, focuses on estimates of land use change in the tropics. Much of the discrepancy in results stems from different definitions of deforestation, which are often based on canopy cover thresholds, or from variations in what is included within or excluded from tropical forests. For example, Achard (2002) considered only the humid tropics (although subsequent studies by Achard et al. (2004) did expand deforestation estimates to include the dry tropics). Other studies may be incomplete to the extent they exclude certain forms of land degradation such as selective logging and fuelwood removals (Fearnside and Laurance 2003; Dixon et al., 1994). Some differences in tropical deforestation rates can be seen in Table 5.

Key challenges in carbon flux estimates stem from a lack of data, and the hazards of aggregating country level data constructed using different underlying methodologies and definitions into one statistic. On top of this, estimation errors from insufficient numbers of permanent data plots may be magnified by the high degree of heterogeneity in tropical forest systems (Dixon et al., 1994). One study suggests that because deforestation in such countries as Bolivia, Columbia, and Peru follow a "clumped" pattern, sample plots must achieve 80% coverage of the forested area in order to reach an accuracy of +/- 20%, compared to FAO data which samples 10% of forest area (Houghton, 2003b). There is also the claim that deforestation rates

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The ability to compare data across regions is difficult due to differences in country-specific definitions of forest cover

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published for the 1980s were actually overstated in national statistics, which has led to underestimations of deforestation in the 1990s (DeFries et al., 2002).

**Table 5** Estimates of tropical deforestation rates

(10 <sup>6</sup> ha/yr)	1980s		1990s		
	FAO (1995)	Defries et al (2002)	FAO (2001)	Defries et al (2002)	Achard et al (2004)
<b>Americas</b>	7.4	4.4	5.2	4.0	4.4
<b>Asia</b>	3.9	2.2	5.9	2.7	2.9
<b>Africa</b>	4.0	1.5	5.6	1.3	2.3
<b>Total</b>	<b>15.3</b>	<b>8.1</b>	<b>16.7</b>	<b>8.0</b>	<b>9.6</b>

Source: Houghton, R.A., 2005b. Chapter 1: Tropical deforestation as a source of greenhouse gas emissions. In Moutinho, P., Schwartzman, S., eds. Tropical Deforestation and Climate Change. Amazon Institute for Environmental Research. Washington, D.C. Reprinted with permission.

### KNOWLEDGE GAPS IN CARBON BUDGETING

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It is important to consider the temporal dynamics of land use change, in terms of historical, current and future impacts on carbon fluxes.

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Beyond the difficulties of determining rates of land use change, there remain many knowledge gaps about carbon fluxes, the location of carbon sinks and sources, and the processes driving them. Many of these gaps have already been mentioned, such as the inadequate number of field measurement plots driving estimates of forest cover and growing stock, particularly in the tropics, and inconsistencies stemming from different accounting methodologies at a national level. As can be seen in Figure 4, understanding the terrestrial carbon cycle not only requires knowledge of the underlying carbon pools, but the fluxes between them, as well as the fluxes into and out of the terrestrial biome.

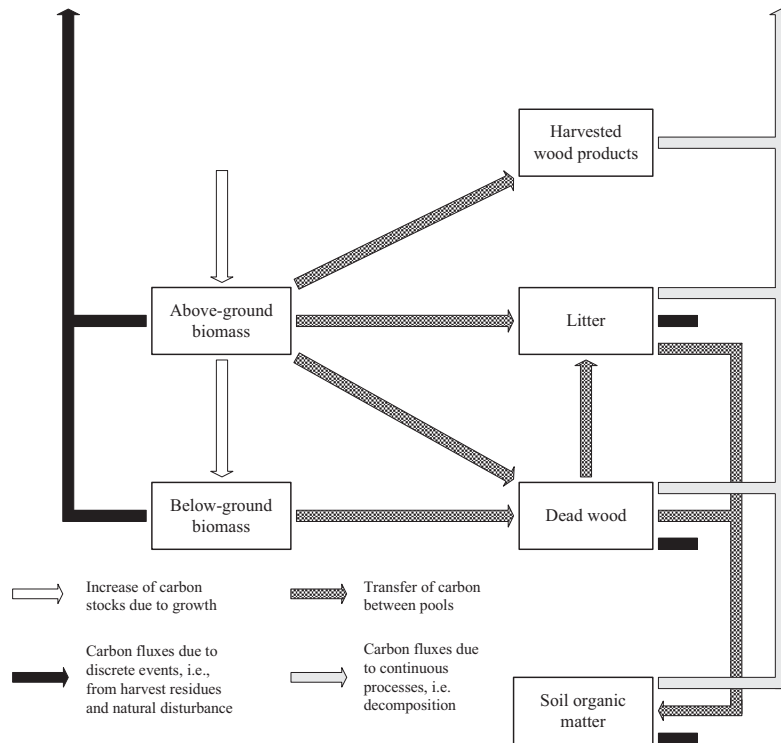
Isolating individual components of carbon pools remains a challenge due to a lack of data. Many studies separate carbon pools into broad categories such as vegetation and soils as seen in Table 6.

The amount of carbon in soils and belowground biomass is not well defined, particularly across forest types (Achard et al., 2004). As discussed in Chapter 2 of this volume, the fact that many soil carbon measurements include only the carbon stored in the first 30 cm may lead to underestimations of belowground carbon pools. However, while this may create challenges for accurate carbon stock measurements, it may be less of a problem for estimates of carbon uptake. Despite the fact that carbon pools may be two to three times higher than aboveground biomass, carbon accumulation in soil is thought to be only 5-15% of forest carbon uptake (Houghton, 2003a).

The amount of biomass in forests and its spatial distribution are also not well known, which is problematic, given that many carbon flux estimates are based on average biomass values (Houghton, 2005a). In measuring biomass, some studies distinguish between forest types within a region (Houghton, 1999; DeFries et al., 2002) while others do not (Brown, 1997; Achard, 2002; Gibbs et al., 2007). Ultimately, actual carbon emissions from deforestation will be driven by the biomass on a

particular site; often these biomass levels may not conform to average values, which can have a significant impact on carbon budgets, particularly in areas with high rates of deforestation. Accurate quantification of biomass figures is also important, not only for a proper accounting of forest carbon stocks, but in determining carbon-related impacts from disturbances such as large scale fires, which can drive interannual variations in measured greenhouse gas emissions (Simmonds et al., 2005).

**Figure 4** Generalized carbon cycle of terrestrial ecosystems showing the flows of carbon into and out of the system as well as between the five C pools within the system



Perhaps the greatest amount of debate, however, focuses on estimates of land use change in the tropics. Much of the discrepancy in results stems from different definitions of deforestation, which are often based on canopy cover thresholds, or from variations in what is included within or excluded from tropical forests.

Source: Intergovernmental Panel on Climate Change (IPCC) 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use, Chapter 2: Generic methodologies applicable to multiple land-use categories, Figure 2.1.

Due to a lack of data availability, carbon flux estimates from different biological processes are often applied across ecosystem types. However, this may create estimation errors since biological processes differ by forest type and thus data may not be transferable. For example, root production is a key component of net primary production, yet accurate data on root dynamics is sparse and often inferred from periodic field measurements of live and dead roots or from biomass estimates from allometric equations (Matamala et al., 2003; Gower et al., 2001; Eissenstat et al., 1997). Researchers caution against applying values from one ecosystem to another and suggest a need for redundant approaches to ensure more accurate estimates (Chapin et al., 2006).

**Table 6 Carbon stock estimates across forest biomes**

Biome	Area (10 <sup>9</sup> ha)		Global Carbon Stocks (PgC) <sup>4</sup>					
	Dixon et al 1994, Atjay et al, 1979	Mooney et al, 2001	Dixon et al 1994, Atjay et al, 1979			Mooney et al, 2001		IGBP <sup>1</sup>
			Plants	Soil	Total	Plants	Soil	Total
<b>Tropical forests</b>	1.76	1.75	212	216	428	340	213	553
<b>Temperate forests</b>	1.04	1.04	59	100	159	139 <sup>3</sup>	153	292
<b>Boreal forests</b>	1.37	1.37	88 <sup>2</sup>	471	559	57	338	395
<b>Tropical savannas &amp; grasslands</b>	2.25	2.76	66	264	330	79	247	326
<b>Temperate savannas &amp; grasslands</b>	1.25	1.78	9	295	304	23	176	199
<b>Deserts and semi deserts</b>	4.55	2.77	8	191	199	10	159	169
<b>Tundra</b>	0.95	0.56	6	121	127	2	115	117
<b>Croplands</b>	1.6	1.35	3	128	131	4	165	169
<b>Wetlands<sup>5</sup></b>	0.35	--	15	225	240	--	--	--
<b>Total</b>	15.12	14.93 <sup>6</sup>	466	2011	2477	654	1567	2221

1) International Geosphere-Biosphere Programme - Data Information Service, Carter et al. 2000, DeFries et al, 1999

2) Estimate likely to be high due to high Russian forest density estimates including standing dead biomass

3) Estimate likely to be high, being based on mature stand density

4) Soil carbon values are for the top 1m, although stores are also high below this depth in peatlands and tropical forests

5) Wetlands not recognized in Mooney et al classification

6) Total land area includes 1.55 x 10<sup>9</sup> ha ice cover not listed in this table.

Due to a lack of data availability, carbon flux estimates from different biological processes are often applied across ecosystem types. However, this may create estimation errors since biological processes differ by forest type and thus data may not be transferable.

Source: Prentice, I.C. 2001. The Carbon Cycle and Atmospheric Carbon Dioxide. Intergovernmental Panel on Climate Change (IPCC) Third Assessment Report, Chapter 3, 183-239.

Lack of temporal-scale data is also a widely recognized problem. Defensible atmospheric measurements did not begin until the 1950s; global coverage was not completed until much later (House et al, 2003). Similarly, there is a wide variety in the availability and extent of historical measurements of forest inventories. To compensate for a lack of historical data, researchers often retrospectively project inventory data by using model simulations based on current data and assumptions about biophysical, climate, and anthropogenic processes over time. Inaccurate estimates of historical inventories, however, can be amplified when used in models to extrapolate future carbon fluxes under different management and climate scenarios.

### MEASUREMENT GAPS IN CARBON BUDGETING

Many knowledge gaps persist because the ability to measure the complexities of forest carbon dynamics over space and time is limited. As mentioned earlier, inventory models that use static data on wood volume to estimate carbon pools and fluxes can result in errors, since wood volume data are principally designed for timber-related analyses. Specifically, the data could be inaccurate, the conversion of wood data to carbon biomass may be incorrect, and extrapolating local data for regional assessments may introduce error (Dixon et al., 1994). Efforts to counter the measurement deficiencies of inventory models include the use of carbon cycle models alongside inventory models to determine net ecosystem production of forests

(Alexandrov et al., 1999). This is thought to create a more dynamic picture of carbon pools and fluxes that encompasses environmental impacts on inventories.

Since carbon pools and fluxes follow a non-normal distribution, bias can result when individual measurements are multiplied to create carbon budgets at a larger scale. Sampling design must accurately reflect the non-normal distribution of carbon pools and fluxes to ensure that carbon “hotspots” are both accounted for and distinguished from the rest of the landscape (Bradford et al., 2009). This is also true for local measurements using eddy covariance methods. Although eddy covariance measurements can be useful in capturing temporal variability of carbon fluxes in areas 1 km<sup>2</sup> or less, it is not considered appropriate to extrapolate these data to draw conclusions on a regional scale or across decadal time frames (House et al., 2003).

Perhaps the largest measurement challenge lies in designing the constraints of a carbon budget model and determining what is and is not included. Different natural and anthropogenic processes drive carbon cycling and have varying degrees of influence across forest types. In temperate forests, precise deforestation rates may not be as critical as determining the appropriate accounting treatment of residual post-harvest organic matter and carbon in wood products, which play a larger role in carbon budgets in these regions (Dixon et al., 1994). In these areas, carbon storage is driven by changes in carbon per unit area as opposed to changes in forested area (Houghton, 2005a). In the tropics, however, accurately measuring land use change is critical since it is a key driver of carbon fluxes. This requires not only a clear definition of deforestation but also clarity as to whether estimates include emissions from other anthropogenic sources such as fragmentation, selective logging, and other types of degradation. It is also important to know whether estimates include all carbon pools, including debris and decaying material (Fearnside and Laurance, 2003). These nuances are not always apparent in published data; yet understanding them is likely to help predict future carbon dynamics in these areas resulting from changing management practices.

Furthermore, estimates of carbon uptake from reforestation may be overstated if reforestation rates are universally applied across all land types. Reforestation of less fertile, more degraded pastures often exhibits slower carbon uptake rates than reforestation of more fertile land. In addition, reforestation estimates must also consider the probability that reforested land may be quickly re-cleared (Fearnside and Guimaraes, 1996).

Finally, measuring the role of disturbance is challenging, since different types of disturbance regimes result in varying patterns in carbon fluxes, both in spatial and temporal terms. Fire results in immediate emissions that can be easily measured through atmospheric-based models. However, insect outbreak and tree mortality may cause more indirect emissions due to the fact that dead organic matter leads to changes in heterotrophic respiration and insect outbreaks may alter age class structure and forest succession (Kurz et al., 2008a). Measurements of disturbance-related impacts must therefore consider the temporal pattern of emissions along with their absolute levels.

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Many knowledge gaps persist because the ability to measure the complexities of forest carbon dynamics over space and time is limited.

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## THE ADDED COMPLEXITY OF CLIMATE CHANGE

The uncertainties in global carbon budgeting due to knowledge gaps, measurement uncertainties, and modelling constraints become even more complex when considered in the context of climate change. Scientists agree that climate change results in both positive and negative feedbacks in forest carbon cycling. Positive feedbacks include greater frequency of fire and tree mortality driven by drought, insect outbreaks, and disease, and reduced albedo from less snow and ice cover. Negative feedbacks include increased forest productivity due to CO<sub>2</sub> fertilization, increased albedo as coniferous forests in northern latitudes transition to deciduous forest types, loss of forest cover due to temperature changes, and more frequent and severe disturbances (Lashoff and DeAngelo, 1997; Betts, 2000).

CO<sub>2</sub> fertilization occurs when increased concentrations of CO<sub>2</sub> in the atmosphere stimulate forest growth thereby increasing the forests' capacity to sequester carbon. Currently, gross photosynthesis of the world's forests is thought to cycle approximately 1/16<sup>th</sup> of the total carbon in the atmosphere and is the main driver of interannual atmospheric variability (Peylin et al., 2005; Prentice, 2001). As long as photosynthetic uptake of carbon remains unsaturated, which is thought to be the case at current atmospheric levels of CO<sub>2</sub>, higher CO<sub>2</sub> levels should lead to increases in net primary productivity (NPP) (Norby et al., 2005). Some researchers point out that increased heterotrophic respiration due to increased temperature will offset a potentially significant portion of this uptake (Cramer et al., 2000). Others challenge the overall role of CO<sub>2</sub> fertilization by suggesting that incremental carbon accumulation attributed to fertilization has actually been driven more by forest regrowth following disturbance than growth enhancement due to CO<sub>2</sub> fertilization (Caspersen et al., 2000).

When measuring increased carbon uptake due to CO<sub>2</sub> fertilization, it is important to know in which pool the increased carbon is stored. If, for example, incremental carbon uptake is stored in stemwood, it will likely be sequestered for a longer period of time than if it goes into fine root production which decomposes rapidly (Norby et al., 2005). In other words, one must consider where the carbon is stored, and not simply NPP rates, in order to understand the effects of climate change-induced CO<sub>2</sub> enrichment on forest carbon fluxes.

Although increased atmospheric levels of CO<sub>2</sub> may stimulate growth of the terrestrial carbon sink, at some point other factors will become limiting, particularly nitrogen, phosphorous and water (Waterhouse et al., 2004; Oren et al., 2001; Schimel et al. 2001; Cramer et al., 2000; Keeling et al., 1996). Research conducted at the Harvard Forest in Massachusetts, USA indicates that if soil is fertile, CO<sub>2</sub> fertilization should drive increased growth in vegetation. On the other hand, vegetation in soils with low fertility may show no response to CO<sub>2</sub> fertilization, while moderately fertile soil may exhibit only short term increases in biomass growth (Barford et al., 2001). Some of this may be offset by increased uptake of anthropogenic nitrogen deposition although the extent to which this can mitigate nitrogen limitation is uncertain (Nedelhoffer et al., 1999). In terms of water

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Although increased atmospheric levels of CO<sub>2</sub> may stimulate growth of the terrestrial carbon sink, at some point other factors will become limiting, particularly nitrogen, phosphorous and water.

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limitation, increased concentration of atmospheric CO<sub>2</sub> initially increases water use efficiency (Cramer et al, 2001). At some point, however, water use efficiency begins to wane, and the benefits of increased CO<sub>2</sub> availability are reduced and ultimately eliminated. Other studies claim that water, nitrogen, and phosphorous limitation have only a small part in reducing CO<sub>2</sub> fertilization effects and that there is significant scope for incremental carbon sequestration through greater photosynthesis (Wullschleger et al., 1995).

Warming temperatures observed under climate change are expected to alter seasonal variations, leading to longer growing seasons. The impact of this on forest carbon cycling is uncertain. Studies in mid- to high-latitudes show that a longer growing season due to temperature increases should increase carbon uptake (Myneni et al., 1997). Chen et al. (2006) found that gross primary productivity (GPP) increased in warmer years while ecosystem respiration (ER) showed lower sensitivity to temperature. Since the difference between GPP and ER is the net carbon uptake, the authors suggest that warmer temperatures will increase carbon accumulation in boreal forests.

However, it is important to consider that warmer temperatures are likely to stimulate thaw in carbon rich peatlands, leading to carbon and methane emissions (Camill, 2005; Camill et al., 2001). Since boreal forests are thought to store 13% of the world's aboveground biomass carbon and 43% of global soil organic matter, understanding the net effects of increased growth of above ground biomass and increases in soil emissions (alongside any expected changes in disturbance patterns) is key to forecasting likely changes in carbon fluxes due to climate change (Chen et al., 2006).

Angert et al. (2005) have further demonstrated the complexity of quantifying climate change effects on carbon by suggesting that in mid- to high-latitudes, decreases in carbon uptake during hotter and drier summers offset increased uptake in the spring, thereby reducing or even eliminating the positive benefits of a longer growing season. In fact, they claim that climate change-induced alterations of regional water balances may be a much stronger driver of carbon fluxes than temperature increases. A zero to small negative effect from a longer growing season is also supported by studies in tropical forests which suggest that decreased photosynthesis in the tropics due to drought effects from warmer temperatures are likely to more than offset any productivity increases in high latitudes due to longer growing seasons (Fung et al., 2005).

Climate change is also expected to alter the natural disturbance regimes of the world's forests, including fire, drought, disease, and insect/pathogen outbreaks. For example, rapid rises in mountain pine beetle induced mortality in Canada are being attributed to climate effects (Logan et al., 2003). Temperature changes, such as lower winter temperatures, allow insects to overwinter while higher summer temperatures (combined with reduced precipitation) have increased water stress (Kurz et al., 2008b). Some examples of insect outbreaks associated with changing temperatures are shown in Table 7.

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Climate change is expected to alter the natural disturbance regimes of the world's forests, including fire, drought, disease, and insect/pathogen outbreaks.

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**Table 7 Insect outbreaks in boreal forests during extreme weather events**

Species	Host(s)	Factors favoring outbreak	Reference
<i>Choristoneura fumiferana</i> (spruce budworm)	Spruce/fir	Warm springs	Ives (1974)
		Moderate winter temperatures	
		Dry years	
<i>Choristoneura pinus</i> (jack pine budworm)	Jack pine	Warm dry summers	Greenbank (1963)
		Drought and high temperatures	MacAloney (1944)
		Below average precipitation	Clancy et al (1980)
		Warm springs	
<i>Malacosoma disstria</i> (forest tent caterpillar)	Aspen	Warm dry periods	Volney (1988)
		Mild winter	Ives (1981)
		Warm springs	

Source: Kurz, W.A., Apps, M.J., Stocks, B.J., Volney, W.J.A., 1995. Global Climate Change: Disturbance Regimes and Biospheric Feedbacks of Temperate and Boreal Forests. in Woodwell, G.M., Mackenzie, F.T., eds. Biotic Feedbacks in the Global Climate System: Will the Warming Feed the Warming? Oxford University Press, New York. Reprinted with permission.

While the net effect of increased disturbance is expected to reduce the efficiency of the global forest carbon sink, climate change impacts on disturbance are not always negative. For example, warmer temperatures may result in increased precipitation in certain regions and decrease fire-related disturbance (Bergeron and Archambault, 1993). In North America, although warmer winters may increase southern pine beetle infestations in North America, they may actually cause decreases in southern areas (Dale et al., 2001). Generally speaking, however, the net effect of climate-related disturbance changes is expected to alter the role of the terrestrial carbon sink by reducing its long term efficiency in sequestering carbon.

Disturbances not only result in emissions at the time of disturbance, but they also alter future carbon fluxes as forests recover from disturbance. Carbon uptake by forests following disturbance is often higher than carbon sequestration on natural, unmanaged stands as young stands sequester carbon at faster rates than mature forests. Forest carbon budgeting must not only account for emission-related disturbance but must have the capacity to incorporate post-disturbance carbon fluxes. However, it would be inappropriate to assume that carbon uptake rates from forest re-growth will exhibit patterns similar to those observed to date. It is important to consider that the compounding effects of higher temperatures, changing precipitation patterns, biodiversity, encroachment of invasive species, as well as northern migration of tree species may alter the composition and growth rate of the vegetation that regenerates in the disturbed area.

In some cases, climate change may cause compounding disturbance patterns. For example, drought-induced water stress may increase a forest's susceptibility to insect outbreak. This may, in turn, lead to higher fuel loads which increase the probability of stand replacing fires (Dale et al., 2001). Understanding the potential second order effects of disturbance patterns in a changing climate is therefore necessary for accurately estimating long term carbon fluxes from natural disturbance.



## IMPLICATIONS FOR POLICYMAKERS

Despite the complexities in measuring forest carbon budgets over time and across different forest types, policies to mitigate climate change must include forests in carbon management strategies. Below are several recommendations to help policymakers navigate the uncertainties:

- Select the carbon budget model that best elucidates those carbon pools and processes under consideration. In other words, atmospheric transport models may be inappropriate when designing policies to manage carbon emissions from land use change. Recognize the constraints on any model used.
- Test the effects of carbon policies against multiple models to ensure against unintended consequences and to better understand the broader impacts of policy design on a wide variety of carbon processes.
- Support the creation of long term research plots to generate better time series data on land use, forest type, and deforestation rates so that long term processes impacting carbon sinks and sources can be better estimated.
- Require methodological consistency in country level accounting of items such as forest definition, cover type, deforestation rates, and biomass conversion factors. Link funding to compliance with global accounting standards.
- Ensure that forest carbon policies consider not only the immediate impacts on carbon sequestration, but the longer term impacts likely to result from policy directives.
- Avoid using region-specific carbon research to make claims about global forest carbon budgets, since model results in one region may not be appropriate in other geographic areas.

## CONCLUSION

Despite the fact that terrestrial carbon cycling is a critical component of the global carbon budget, it is probably the least understood and most widely debated. While models exist to measure carbon pools and fluxes in terrestrial ecosystems, no model is able to fully account for all the natural and anthropogenic processes driving carbon fluxes across time and space. When estimating climate change impacts on forest carbon, it is extremely difficult to aggregate the positive and negative feedbacks due to changes in temperature, moisture, and disturbance patterns.

Nevertheless, new forest management strategies must be designed to better optimize terrestrial carbon storage capacity while protecting long term forest carbon sinks from the effects of climate change and human-induced land change. This will require continued research to better measure global forest carbon fluxes while

encouraging a global effort to amass consistent and comprehensive data on the current state of forests across the earth. Although it may be a difficult undertaking, it will be necessary to ensure the ongoing health of the world's forests and their ability to continue as a critical carbon sink.

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## Part II: The Management of Carbon in Forests and Forest Products

### SECTION SUMMARY

The following five papers provide a comprehensive synthesis and review of carbon management in forests and the life cycle of associated wood products. The papers highlight areas of what is known from recent research, and where the gaps are.

The first three papers discuss the management of forest carbon in existing tropical, temperate, and boreal forests, and in afforestation and reforestation projects. In temperate and boreal forests resiliency treatments (such as fuel reduction thinning and prescribed fire) result in lowered vegetative carbon storage, but they help produce forests that are significantly less susceptible to catastrophic disturbance (with accompanying drastic carbon release). In the tropics, reduced impact logging (RIL) is an important practice to lessen carbon loss, but it is necessary to move beyond RIL to substantially increase carbon storage by developing a more sophisticated silviculture. The largest potential source of carbon sequestration in the tropics is the development of second growth forests on old agricultural lands and plantations established on appropriate sites. However, for all forests, the risk of leakage must be addressed. If carbon sequestration strategies simply displace timber harvests from one forest to another, the ultimate carbon gain is questionable.

The final two papers evaluate post-harvest strategies of carbon management. Some studies find that substitution of wood for other construction materials (e.g., steel and concrete) produces net GHG emissions reductions. Substitution effects may be up to 11 times larger than the total amount of carbon sequestered in forest products annually. However, paper products contain significantly more embedded fossil fuel (carbon) energy than wood products, and newer wood products such as oriented strand board and laminated veneer lumber use 80-216% of the energy needed to produce solid sawn lumber. The end-of-use pathways of wood products are integral to the carbon cycle. Once discarded, wood products can be burned for energy production, recycled or reused, or put in landfills, where the carbon can remain indefinitely due to anaerobic conditions.

Contributors toward organizing and editing this section were *Mark S. Ashton, Deborah Spalding, Thomas Graedel, Mary Tyrrell and Reid Lifset.*

## Chapter 9

# Managing Carbon Sequestration in Tropical Forests

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### EXECUTIVE SUMMARY

This chapter examines how management methods can be implemented to reduce carbon loss and increase carbon storage in tropical forests. Tropical deforestation and degradation are contributing 17 percent of total annual global greenhouse gas emissions. As policy makers work to develop solutions that address climate change, there has been considerable focus on incorporating tropical forests into the overall climate solution. Silvicultural practices will need to be an integral part of reducing carbon loss and improving carbon storage if we are to solve this global challenge while meeting resource needs. The following are important considerations highlighted in this chapter:

- Global climate change negotiations have begun to focus on sustainable forest management as a means to achieving carbon emission reductions, thus presenting opportunities in tropical forest management.
- The most important goal in managing tropical forests for carbon is to conserve standing forests, especially primary forests that are high in carbon.
- Forest carbon storage and uptake vary significantly based on climate, soils, hydrology, and species composition. It is necessary to consider these factors when managing a tropical forest for carbon.
- Reduced impact logging (RIL) is an important practice to lessen carbon loss, but it is necessary to move beyond RIL to substantially increase carbon storage by developing more sophisticated, planned forest management schemes with silvicultural treatments that ensure regeneration establishment, post establishment release, and extended rotations of new stands.

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Some work on silviculture has been done in the rainforest regions (ever-wet and semi-evergreen), but only in very specific places; almost none has been done in montane or seasonal (dry deciduous) forests.

- Forests can be financially viable compared to other land uses through integration and cultivation of species that provide timber and non-timber products that are stacked (cumulative) and that are compatible with service values – carbon sequestration and water quality.
- To increase forest carbon storage while also meeting societies' resource needs, it is essential to engage in stand- and landscape-level planning aimed at increasing carbon storage.
- Many logged over and second growth forests are ideal candidates for rehabilitation through enrichment planting of supplemental long-lived canopy trees for carbon sequestration.
- The largest potential source of carbon sequestration in the tropics is the development of second growth forests on old agricultural lands and plantation systems that have proven unsustainable. Every incentive should be provided to encourage this process.

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The implementation of basic forest management methods throughout the tropics has the potential to considerably reduce carbon loss and increase carbon uptake and storage.

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**Keywords:** silvicultural practices, reduced impact logging, carbon sequestration, emission reductions, avoided deforestation, forest biomass, carbon stocks, REDD+, climate change policy

## INTRODUCTION

Tropical forest systems play a large role in the global carbon cycle, with tropical vegetation and soils holding almost 17 percent of all carbon stored in terrestrial ecosystems (Schlesinger, 1997). The large size and distribution of tropical forests make them a significant carbon reservoir (Schimel, 1995). It has been calculated that tropical deforestation and degradation currently account for 20 percent of the annual global carbon emissions (IPCC, 2007) that are contributing to global climate change. In this chapter, we discuss the management options available to help conserve the carbon stored in tropical forests while also reducing the loss of carbon due to forest degradation and removal.

Tropical forests around the world are undergoing a dramatic change, with primary forests (high in biomass and carbon) being converted to agricultural lands or degraded forests (low in biomass and carbon). A significant proportion of this change is occurring as a result of logging intensification throughout the tropics (Pinard et al., 1995; Uhl and Kauffman, 1990, Verissimo et al., 1992). Logging operations in the tropics rarely use best forest management practices or silvicultural methods and this has resulted in extensive loss of carbon (Putz and Pinard, 1993). The implementation of basic forest management methods throughout the tropics has the potential to considerably reduce carbon loss and increase carbon uptake and storage

(Putz et al., 2008). Efforts to maximize carbon uptake and reduce carbon loss need to be based on site dynamics and well-planned management practices, as well as the application of silvicultural practices that are based on forest type and site characteristics. The high levels of carbon loss that result from deforestation and degradation have led to a considerable focus on incorporating tropical forests into the overall climate mitigation solution (IPCC, 2007). Forest management policies and silvicultural practices are and will continue to be an integral part of efforts such as REDD+ that are aimed at reducing carbon loss and improving carbon storage.

This chapter will provide an overview of tropical forest management practices and how they can be used to manage tropical forests for carbon. First we will present and outline the major tropical forest biomes and discuss how carbon is related to forest type and the site characteristics of each region. We will then discuss several key concepts that are important to understand when it comes to managing tropical forests for carbon. Then we will discuss the practice of reduced impact logging (RIL) and how it can be applied to reduce the carbon lost through conventional logging practices. After discussing RIL, we will shift our focus to management practices and silvicultural treatments that are rarely used now, but which could significantly improve carbon storage and uptake in tropical forests. Finally, we will end with a summary of the key findings and policy implications that are outlined within the chapter. It is our hope that this will be instructive for land managers and policy makers who are seeking to better understand the various approaches that are available and appropriate for managing tropical forests for carbon.

## **TROPICAL FOREST BIOMES**

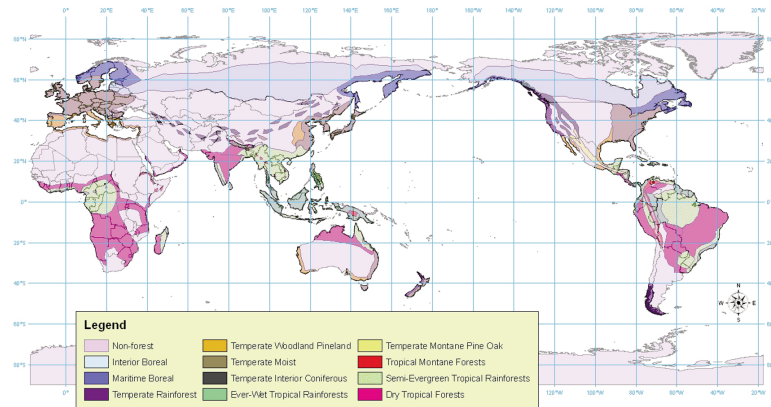
Tropical forests are found throughout the equatorial regions of the world and are broadly categorized by region: Africa, the Neotropics (Central and South America), and Asia (Figure 1). Each region can be divided into three major forest biomes: tropical rainforests (ever-wet, semi-evergreen), tropical montane forests, and tropical seasonal forests (dry deciduous). All biomes are loosely contained between the Tropic of Cancer (23°N) and the Tropic of Capricorn (23°S) and encompass a broad range of regional expressions that vary based on elevation, soil conditions, and regional climatic variations. Carbon uptake and storage vary significantly across biomes, with an average of 200 Mt C/ha in tropical rainforests and 140 Mt C/ha in tropical seasonal forests (Houghton 1999; DeFries et al., 2007). Within each region (Asia, the Neotropics, or Africa) similar biomes can have dramatically different carbon values, which also fluctuate across the landscapes of these regions (Figure 2) (IPCC, 2006).

### **Tropical rainforest**

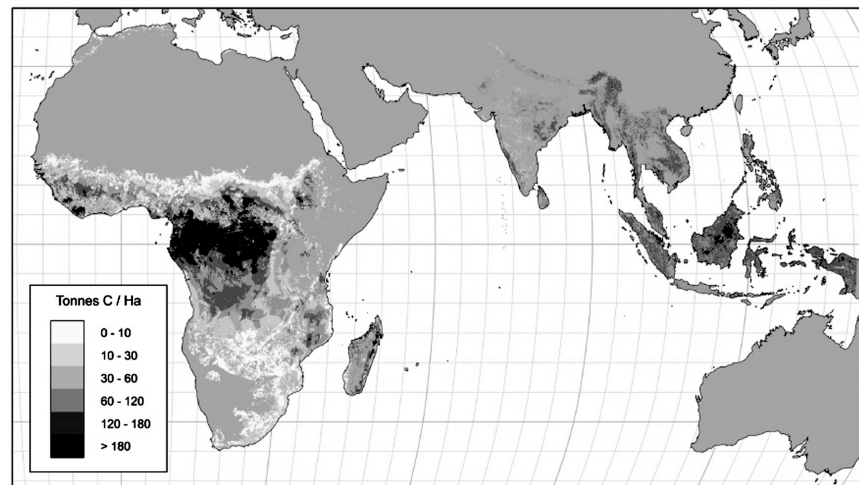
Tropical ever-wet and semi-evergreen forests are characterized by more than 80 inches (2,000 mm) of rain annually. These forests have the highest vegetation biomass as well as the largest carbon stocks of all tropical forests (Holzman, 2008). The most species rich and structurally diverse forests are around the equatorial

latitudes. The greatest expanses of semi-evergreen rainforests are found in the Amazon Basin and the Congo Basin, while the greatest expanses of ever-wet rainforests are on the Southeast Asian islands of Borneo, Sumatra, and New Guinea. The three regional expressions of the tropical rainforests – Neotropical (Central America, the Pacific coast of northern South America, the Amazon, the Caribbean), African (West Africa, Central Africa), and Asian-Pacific regions (South Asia, Indochina, maritime Southeast Asia, Australia/New Guinea) – are each distinct from one another in terms of forest tree composition as well as in carbon levels. These differences are the result of biogeographical origin, climate, soil, and forest structure (Holzman, 2008). The greatest similarity exists between the Amazon and Central Africa forests because of a common, but ancient biogeography.

**Figure 1** Original extent of boreal, temperate, and tropical forest types of the world prior to land clearing



**Figure 2** Forest biomass carbon maps for Africa and Southeast Asia produced by using regression-based models to extrapolate forest inventory measurements



Source: From Gibbs, H.K., Brown, S., Niles, J.O., Foley, J.A. 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters* 2, 045023 (13 pp). Reprinted with permission.



### *Regional variations*

The neotropical rainforest of Central and South America is the largest and most extensive of the tropical rainforest biomes. A number of tree families are represented in the canopy layer of these forests: Brazil nut (Lecythidaceae), the genera *Tabebuia* (Bignoniaceae), *Anacardium* (Anacardiaceae), and *Vochysia* (Vochysiaceae), and many genera (e.g. *Parkia*, *Cedrelinga*, *Dalbergia*, *Dipteryx*) in the Leguminosae (Meggers et al., 1973). The single most important timber family in the neotropics is mahogany (Meliaceae) with genera such as *Guarea*, *Swietenia* and *Cedrella* dominating the timber markets. The Neotropical forests, which currently store between 120 – 400 Mt C/ha depending upon species composition, soil, and climate, have historically and continue to experience high levels of deforestation (IPCC, 2006).

The African rainforest is smaller in size with less species diversity than the other regions. Trees tend to be shorter, and the forest less dense, with levels of carbon ranging from 130 – 510 Mt C/ha (IPCC, 2006). Within the African tropical forest, the canopy layers tend to consist of members of the Caesalpinioideae subfamily of the legume family and include *Gilbertiodendron*, mopane (*Colophospermum mopane*), and senna (*Senna siamea*) (Meggers et al., 1973). However, the most important timber family in this region is again in the Meliaceae, represented by the African mahogany genera (*Entantrophragma*, *Khaya*).

The Asian-Pacific forest is distinctive due to the presence of the Dipterocarpaceae family of trees that dominates the forest composition. These trees are among the tallest in the tropical rainforest biome and occur in large clumps (Holzman, 2008). It is the dominance of the Dipterocarpaceae tree family that gives these forests the highest carbon levels (120-680 Mt C/ha) (IPCC, 2006), along with the high carbon peat swamps (>1000 Mt C/ha) of the region.

### *Lowland rainforests*

Lowland tropical forests exist below 300 meters elevation and constitute the vast majority of tropical ever-wet and semi-evergreen rainforests. These forests have a diversified forest canopy system with the greatest number of commercial tree species, such as the dipterocarps (Dipterocarpaceae), Brazil nut (*Bertholletia excelsa*), and mahogany (*Swietenia*, *Entantrophragma*, *Khaya*, *Cedrella*) species. Lowland tropical forests comprise most of the lower Amazon and Congo Basins. On their outer margins and along the major river ways, they are being logged and converted at a much faster rate than other tropical forests, since their soils are more suitable for agriculture and the land more accessible (Fearnside, 1993). Within the lowland rainforests, the peat swamps, where elevated water tables inhibit decomposition, have a substantial storage of organic matter. Draining these peat swamps results in a significant loss of stored carbon (Dixon et al., 1994).

Coastal lowland rainforests are usually located on fertile soils and tend to be extremely workable and in areas of high human influence (Fearnside, 1993). Coastal forests usually have flat, deep soils with a sandy component, making them suitable for plantations and tree crop agriculture systems such as oil palm, rubber, and coconut species (Ashton, 2003). As a result, many of these forests have been cleared and

replaced with tree crop agriculture that requires intensive inputs. These forests will likely remain in this state given their productivity and proximity to markets. As a result, relatively few coastal rainforests still exist. Where they do persist, as in the Chocó rainforest along the Pacific Coast of Panama and Colombia, they hold a large amount of stored carbon (Leigh, 1999).

### *Hill rainforests*

Inland, rainforests with elevations greater than 300 m are much more variable and diverse, with major differences in stored carbon between the broad flat areas and the hilly uplands. These forests are often on marginal lands, in terms of fertility, since the soils often have higher clay content and poor structure (Schimel, 1995). The steep slopes in these areas make them less workable and more prone to erosion. When these forests are converted to agriculture and range land, which has occurred in many regions, they are more likely to be abandoned over time and to revert to secondary forests. Often, these forested hilly regions are part of important catchments that provide water for coastal cities and for irrigating crops in coastal lowlands (Ashton, 2003). This combination of factors makes these forested regions ideal for long-term carbon management as well as for co-benefits like water.

### **Tropical montane forests**

Tropical montane forests grow above an altitude of 1000 meters. For wet tropical rainforests, an increase in altitude results in changes in forest structure (Vitousek and Stanford, 1986). Primarily, these forests become shorter, thicker, and denser with a less developed canopy strata system. They tend to hold less carbon on average than lowland tropical forests as a result of the slower growth rates. Despite this, increased precipitation and decreased decomposition have led to high soil carbon levels, with as much as 61.4 Mt C/ha found in montane forest regions in Ecuador (Rhoades et al., 2000). Regional comparisons show that montane tropical forests in Africa hold between 40-190 Mt C/ha, in the Neotropics 60-230 Mt C/ha, and in the Asia-Pacific region, where the highest carbon stocks have been recorded, 50-360 Mt C/ha (Gibbs et al., 2007). South America holds the majority of montane forests because of the Andes, whereas in Africa they are restricted to the upper slopes of the volcanic island mountain systems of East and Central Africa.

Soils in montane forests are usually very fertile, consisting of inceptisols or histosols. As a result, montane forests located adjacent to populated areas often experience a significant loss of forest and soil carbon to intensive agriculture (market gardens – vegetables). Because of the steep slopes, they are also easily eroded. Rhoades et al. (2000) found almost a 24% decrease in soil carbon levels between primary forests and sugar cane fields at high elevations in Ecuador due to soil disturbance and soil erosion. When montane forests are less accessible and in unpopulated regions, these pressures are less intensive. Isolation makes these forests more suitable for carbon reserves because they are less threatened by agricultural conversion and timber extraction (see Chapter 15, this volume, for further discussion on this topic)

## Tropical seasonal forests

Tropical regions with distinct seasonal rainfall are home to dry deciduous forests. These forests are found in wide bands along the perimeter of the Tropical Rainforest biome towards the margins of the tropical latitudes between 10° and 20° N and S latitudes (Holzman, 2008). Seasonal tropical forests primarily occur in South Asia, West and East Africa, northern Australia, the Pacific side of Central America, eastern Brazil, and the southern rim transition region of Amazonia. In seasonal forests there is a distinct cooler and extended dry season and a distinct wet season with precipitation less than 2,000 mm per year unless the climate is strongly monsoonal, where rainfall can be very high but over a short time interval, making most of it surplus. On average these forests tend to be less diverse and more dwarfed in terms of tree size. Fire and large ungulates can play an important role in regulating forest understories in comparison to typical equatorial rainforests. Many of the same families of trees found in tropical rainforests are also found in seasonal tropical forests: however, the species are quite different (Holzman, 2008). Trees in the fig family (Moraceae) are widespread throughout all regions, as are trees in the kapok family (Bombacaceae) such as kapok (*Ceiba pentandra*) and palo barrocho (*Chloroleucon chacoense*) trees in the Neotropics and baobab (*Adansonia*) in Africa (Bullock et al., 1995). Many legumes in the subfamilies Mimocaceae (e.g. *Albizia*, *Acacia*) and Fabiaceae (e.g. *Gliricidia*) are common in both the Neotropics and Africa. The seasonally dry miombo woodlands that create an arc around the wet evergreen forests of Central Africa are dominated by *Brachystegia* (Caeasalpinioideae). Trees in Asian seasonally dry forests are often represented by species in the Combretaceae (*Terminalia*), Verbenaceae (teak – *Tectona*, *Vitex*), Ebenaceae (ebony – *Diospyros*) and Dipterocarpaceae (sal – *Shorea robusta*) families (Bullock et al., 1995).

As with tropical rainforests, there are distinct regional differences that exist in tropical seasonal forests in terms of species composition, soil quality, and climatic variables, all of which affect the levels of biomass and carbon storage found within each region. This is evident in the regional carbon variations that exist within seasonal tropical forests, with Africa having on average 140 Mt C/ha, the Neotropics 210 Mt C/ha, and the Asian-Pacific region holding the lowest of all, 130 Mt C/ha.

Under normal climatic conditions, major fires do not appear to be a frequent occurrence in seasonally dry tropical forests (Malaisse, 1978; Hopkins and Graham, 1983). The most vulnerable dry forests have been found to be those adjacent to savanna vegetation because of the sparseness of ground vegetation under the forest canopy (Hopkins and Graham, 1983). Malaisse (1978) found that local people started most fires in the African miombo (woodland) ecosystems during the dry season to maintain the areas for grazing. Thus, when managing a tropical dry forest for carbon, it is vitally important to work with local people to reduce the risk of fire in these forests and develop solutions that work well with local needs.

Another general characteristic of tropical seasonal forests is that their soils overall are more fertile than in wet tropical regions. Forests of this type have therefore received proportionally greater impact from land conversion to agriculture. More so than in wet evergreen regions, human populations are higher in number, but are

often poorer and are dependent on fuelwood from the forest. Such forests are now restricted to the most marginal lands and represent a small fragment of what they once were. All of this again emphasizes the importance of site and regional knowledge in managing these forests for carbon.

### KEY CONCEPTS

In order to fully understand how tropical forests are affected by forest management practices, it is important to understand a few key concepts.

#### Primary tropical forest vs. managed tropical forest vs. second growth

Primary tropical forests are forests that have attained a great age and exhibit a structural variety that provides higher habitat diversity than forests in other stages. Primary forests usually have multiple horizontal layers of vegetation representing a variety of tree species, age-classes, and sizes (Clark, 1996). This structural complexity, combined with the long-term accumulation of litter and debris in the soil layers, results in high carbon storage levels within primary forests.

Managed tropical forests are forests where there is an effort to maximize desired values (timber, carbon, water, biodiversity) and reduce unwanted attributes. A managed forest will not contain necessarily as much carbon as a primary forest; thus, any activity that converts a primary tropical forest into a managed forest will likely result in carbon loss.

Secondary tropical forests are forests that have grown after a significant disturbance such as fire, insect infestation, timber harvesting, or from land clearance for agriculture. Secondary tropical forests tend to have smaller trees in the stand initiation or stem exclusion phase of stand development and will typically lack the large, high carbon, late stage canopy trees (see Chapter 3, this volume, for a detailed discussion of stand developmental stages and carbon). As a result, these forests hold less carbon than primary tropical forests (Brown and Lugo, 1990). Secondary forests are common in areas where forests have been cleared for other land uses like agriculture and were later abandoned, as is the case for large areas of Panama and Costa Rica. As secondary forests grow, they exhibit high levels of carbon uptake and are generally considered significant carbon sinks (Brown and Lugo, 1990).

#### Maximizing carbon uptake vs. maximizing carbon storage

Carbon uptake is maximized in tropical forests during the initial stand developmental stages when biomass productivity is at its greatest. The rate of carbon uptake will slow with time as growing space is occupied. In comparison, maximum carbon storage is achieved in the later stages of stand development when a large amount of carbon is stored in canopy trees (Kirby and Potvin, 2007). Older forests, with well developed stand structures, will also have higher soil carbon levels than forests in earlier developmental stages (Kirby and Potvin, 2007).

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The rate of carbon uptake will slow with time as growing space is occupied. In comparison, maximum carbon storage is achieved in the later stages of stand development when a large amount of carbon is stored in canopy trees

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Thus in managing tropical forests for carbon, it is important to determine if the forest is going to be managed for maximum carbon uptake or maximum carbon storage, since this determines management practices over time (Kirby and Potvin, 2007).

### **Site and climatic factors limit productivity and carbon storage potential**

Forested landscapes in the tropics vary greatly in terms of biomass productivity and capacity for carbon storage and uptake and, as a result, forest managers will need to take into account all site characteristics across the landscape to assess carbon uptake rates and the carbon storage potential. These differences can be observed at the regional scale (average carbon biomass estimates given in IPCC, 2007). On a more local landscape scale, soil fertility, precipitation levels, and disturbance regimes all greatly influence the maximum amount of biomass and carbon that can exist at a location (Gibbs et al., 2007). Tropical forest soils, such as oxisols and ultisols, tend to be deeply weathered and have little to no organic or humus layer (using the USDA (1975) soil classification). In some areas, such as in montane forests, the soils are younger and of volcanic origin, making them fertile and desirable for agriculture. Younger soils, such as inceptisols and entisols, occur on alluvial plains and along rivers or at their ends as deltas and are extremely productive, whereas others are representative of nature's erosive forces (landslides) (Holzman, 2008). Tropical forest managers can manipulate forests to adjust carbon uptake levels or manage for species compositions that contain large amounts of carbon, but they will not be able to produce more carbon storage than the site is capable of unless they add fertilizers, add water or take water away, and this is usually too costly for land that is marginal – which most primary forests are now restricted to.

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In managing tropical forests for carbon, two approaches are possible: create carbon additionality or minimize the carbon lost from forest management activities.

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### **Creating carbon additionality vs. minimizing carbon loss**

In managing tropical forests for carbon, two approaches are possible: create carbon additionality or minimize the carbon lost from forest management activities. To create additionality, the forest management practices and methods must increase the amount of carbon held within forests (Lugo et al., 2003). Reforestation, when forests are planted on degraded lands, would be an example of additionality. Alternatively, forest management practices can be implemented that minimize the amount of carbon lost in comparison to a set of baseline management conditions. This form of carbon accounting is the basis for using reduced impact forest management practices as a means to minimize carbon loss from current levels.

### **Forest degradation**

In order to best determine the appropriate silvicultural treatment to maximize carbon uptake and storage in a forest, it is first necessary to identify the nature of the disturbance and the type of degradation affecting the site. This requires an understanding of forest degradation processes, since so much of the tropical forest biome would now be considered second growth, logged over, or re-growth on old

agricultural lands (Ashton et al., 2001a; Chazdon, 2008). Degradation processes can be divided in two categories, structural and functional (Ashton et al., 2001a).

*Structural degradation* is caused by disturbance regimes that alter species composition, structure, and regeneration of a forest. Disturbance regimes that promote structural degradation can be chronic (either bottom up or top down – see Figures 3a and 3b), or sudden and acute such as partial land clearance for agriculture (swidden systems) or one time intensive logging.

Chronic bottom-up impacts occur when the understory strata of a forest is continually suppressed. As a consequence, the forest structure is simplified because the lower strata lose their ability to successfully regenerate. Examples of such processes are the continuous presence of ungulates and associated herbivory or the intensive cultivation of non-timber forest crops in the understory. The forest becomes impoverished of understory shrubs and tree species as well as seedling regeneration of canopy trees.

Chronic top-down impacts occur when disturbances directly affect the forest canopy. An example would be selective logging with repeated diameter-limit cutting at frequent intervals that progressively removes the tallest trees in the canopy. Here composition and structure shifts from dominance of the large timber tree species to tree species of the subcanopy and the understory.

Land suffering acute impacts is land that has been partially cleared for agriculture and remains in cultivation for only a short period (less than 5 years) before it is abandoned. After abandonment, the site is colonized by pioneer species and sprouts from stumps and root suckers. The biggest shortfall is the absence of late-successional species that have been eliminated permanently from the site because their advance regeneration was eradicated by cultivation (Figure 3c).

#### **Top down disturbance**

*An example of top-down disturbance is the diameter-limit cuttings that target individual trees in periodic cycles of 10 to 30 years. At the beginning, the effects of such disturbance can be considered harmless, but over time the canopy will progressively lower in stature and subcanopy tree species and vines will occupy the upper stratum. With the removal of the late-successional canopy trees, the seed source for these species also disappears. This causes loss of advance regeneration and the simplification of forest stratification from the top downward (Ashton et al., 2001a).*

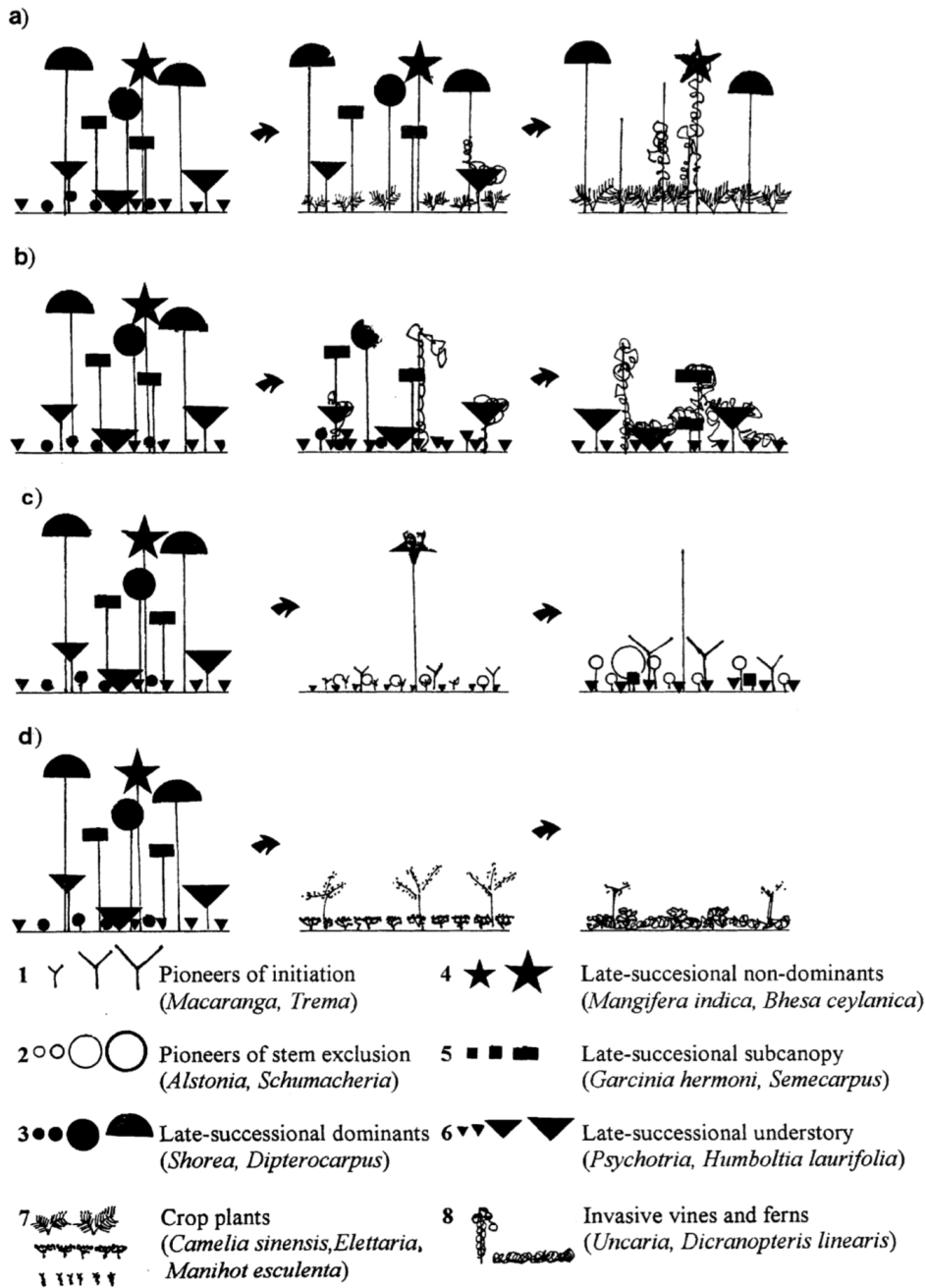
*Functional degradation* is caused by acute disturbances that are severe and lethal to the groundstory such that it is eradicated. In most cases the soil is intensively turned over with the roots and stumps being removed. Such impacts go beyond shifting forest structure and composition to permanently affecting soil fertility and structure, then altering infiltration, water holding capacity, and therefore subsurface hydrology. The disturbance usually associated with functional degradation is forest clearance for intensive agricultural cultivation or permanent conversion for development (Figure 3d) (Ashton et al., 2001a).

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In order to best determine the appropriate silvicultural treatment to maximize carbon uptake and storage in a forest, it is first necessary to identify the nature of the disturbance and the type of degradation affecting the site.

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**Figure 3 Stand development profiles depicting degradation chronosequences of stand composition and structure for a mature mixed dipterocarp forest:** a) bottom-up effects of structural degradation; b) top-down effects of structural degradation; c) functional degradation from a one-time incomplete clearance; d) functional degradation from a one-time complete clearance.



Source: Ashton, M.S., Gunatilleke, C.V.S., Singhakumara, B.M.P., Gunatilleke, I., 2001a. Restoration pathways for rain forest in southwest Sri Lanka: a review of concepts and models. *Forest Ecology and Management* 154, 409-430.

### Acute intensive and prolonged disturbance

*Permanent conversion of forest to alternative land uses often leads to functional degradation. Of all disturbances, this has the most detrimental effects on soil erosion, hydrological regimes, and edge effects. The majority of tree species' establishment processes (advance regeneration, vegetative sprouting) have been eliminated from the site. After abandonment, the site usually transitions into non-forest composed of fire-prone grasses and ferns. Many of these colonizers are exotic and/or invasive. Once these species have established, they tend to self-perpetuate because of their root networks and their ability to quickly regenerate after fire (Ashton et al. 2001a).*

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In many areas, issues such as land tenure, the lack of environmental regulations, or the inability to enforce existing environmental laws exacerbate or encourage inappropriate logging methods and discourage sustainable forestry practices. As is often the case, there is also a lack of trained experts available to provide guidance in appropriate forest management techniques.

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### CARBON IMPACTS ON TROPICAL FOREST MANAGEMENT PRACTICES

At the foundation of all tropical forest management is the need to create additionality by reducing the loss of primary tropical forests, which hold the vast majority of the carbon in all types of forest. Minimizing the amount of carbon lost beyond a set baseline by improving logging practices is an important aspect of forest carbon management, but in the end, carbon is still being lost, just not as much as before. Once large commercial trees, high in biomass and carbon, have been logged out of primary tropical forests, the aboveground and below ground carbon levels will take multiple decades to reach maximum carbon storage again.

In order for tropical forest management practices to truly create additionality, they need to occur on lands where the rehabilitation of tropical forests can occur and carbon can be added (IPCC, 2007). This can occur through the establishment of managed tropical forests that provide multiple values on previously degraded lands. The goal of forest management practices should be to provide additional carbon uptake and storage while reducing carbon loss.

Throughout much of the tropical region, forest management is in the initial stages of development. South Asia is an exception, where the British colonizers established forestry in the mid-1850s (Ashton et al, 2001a; Ashton, 2003). Generally speaking, very few tropical forests are managed using silvicultural treatments. This is the result of a number of destabilizing social phenomena that deter investments in basic forest management practices (Uhl, et al, 1997). In many areas, issues such as land tenure, the lack of environmental regulations, or the inability to enforce existing environmental laws exacerbate or encourage inappropriate logging methods and discourage sustainable forestry practices. As is often the case, there is also a lack of trained experts available to provide guidance in appropriate forest management techniques. In addition, other factors such as access to markets and the lack of financial incentives to implement improved forest management practices drive forest management decisions in these often-impoverished areas.

Management practices developed for the tropics rarely encourage long-term carbon storage (Putz, et al., 2008). Silvicultural treatments are usually aimed at improving the timber production of commercial species such as mahogany and



dipterocarp, or for various non-timber forest products (Feldpausch, et al, 2005). Seldom do these treatments favor carbon sequestration. The thinning or harvesting of undesired tree species will lower carbon yields over the short term and often limit carbon sequestration in the long term. In the following section we outline the range of silvicultural treatments that are available for tropical forest management, starting with the most basic and moving towards the more sophisticated.

### **Reduced Impact Logging (RIL)**

A total area of 350 million hectares of tropical forest is designated as production forest (ITTO, 2006). These forests, high in biomass, are primarily used for the extraction of commercial timber to supply growing domestic and international markets. Under conventional logging practices, for every tree logged, 10 to 20 other trees are severely damaged by untrained fellers and machine operators working without the aid of detailed maps or supervision (Sist and Ferreira, 2007). Carbon lost from this damage can be extensive, with 30-40 percent of the area often affected by heavy equipment (Chai, 1975; Jusoff and Majid, 1992).

Reduced impact logging (RIL) refers to the use of improved harvesting and forest management practices, in combination with education and training, to reduce avoidable logging damage to residual forest, soils, and critical ecosystem processes (Pinard and Putz, 1996). In well-managed forests in the developed world, this is normally standard practice and implemented by the majority of forestland managers. The practice of RIL is not technically a silvicultural treatment, and in the tropics RIL is not required and rarely employed (Putz et al., 2000).

Nonetheless, the use of RIL has the potential to significantly reduce the carbon losses associated with conventional logging. With fewer trees killed or damaged (Johns et al., 1996; Pinard and Putz, 1996) more carbon remains in the living forest. If these residual trees are of higher diameter classes, then a larger amount of carbon will remain sequestered and there is a greater potential for future carbon storage (Johns et al., 1996). Soil carbon is often a significant proportion of the carbon lost due to conventional logging. Forests subject to conventional logging lose much of their silvicultural value due to soil damage (Putz and Pinard, 1993). As a result, reducing soil damage is a major emphasis of RIL, especially where logging operations occur on steep slopes and use heavy machines on wet soil. These practices significantly disturb and erode soil and release stored carbon (Putz et al., 2008a).

Potential carbon reductions from improved harvesting practices are often significant. Research in Southeast Asia has shown that RIL areas contain more than 100 Mg more biomass per hectare than conventionally logged areas one year after logging. (Pinard and Putz, 1996). Given the large areas of tropical forest designated as production forests around the world, the implementation of RIL provides a significant opportunity to reduce carbon emissions from tropical forests.

Extensive research over the past three decades has provided the scientific grounding for the development of RIL guidelines, outlined in Table 1 and condensed from the FAO Model Code of Forest Harvesting Practices (Dykstra and Heinrich, 1996). These pre- and post-logging guidelines are designed to retain forest biomass

and protect future regenerative growth, thus reducing carbon loss and providing the foundation for future carbon sinks as regeneration fills the growing space.

**Table 1 Reduced-impact logging planning and harvesting guidelines Condensed from FAO Model Code of Forest Harvesting Practices**

Harvest Plan	Formal plan prepared based on timber stock and locations of commercial trees, proposed roads, skid trails, stream crossings, buffer zones, logging unit boundaries
Pre-felling vine cutting	All vines >2 cm DBH to be severed at least 12 months prior to harvesting
Skid trail planning	Skid trails to be located on ridges and designed to minimize skidding distances, skidding on steep slopes, skidding downhill, and stream crossings
Tree felling	Decisions on felling directions based on safety to feller, ease of skidding, and avoidance of damage to harvested tree

Source: Adapted from Dykstra and Heinrich (1996)

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In most tropical rainforest regions, silvicultural knowledge is very varied – with South Asia and Australia having high knowledge and almost all other regions having a depauperate knowledge.

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## **BEYOND REDUCED IMPACT LOGGING: SUSTAINABLE TROPICAL FOREST MANAGEMENT**

It is important to remember that RIL practices can still result in significant loss of stored carbon. To move beyond RIL, and produce greater carbon gains than those obtained from RIL, it is important to look at how forest management practices can be used to increase carbon uptake and storage. To do this, forest managers need to carefully take into consideration a landscape's underlying soil and hydrology as well as the disturbances that are acting on the forests (Ashton et al., 2001a; Ashton, 2003). In most tropical rainforest regions, silvicultural knowledge is very varied – with South Asia and Australia having high knowledge and almost all other regions having a depauperate knowledge. Silvicultural information for montane and dry deciduous forests is almost completely lacking.

### **Forest management and planning**

#### *Scaled land use planning*

To effectively store carbon over the long term, it is important for forest managers to delineate forest stands into protected and production stands based on desired forest values, including carbon. The dipterocarp forests of Asia have been heavily logged and the remaining forests are now in the hills and mountains of the region. These areas are recognized by governments as catchments for drinking water and are also ideal for the long-term sequestration of carbon (Ashton, 2003).

This example highlights how land management planning can help to ensure long-term carbon storage while meeting society's resource needs. Engaging in landscape scale management based on functionality ensures that the maximum amount of carbon can be sequestered without compromising the long-term sustainability of the carbon storage.

The use of stands as the management unit within tropical forests has been largely disregarded in favor of large scale and broadly applied management prescriptions (Appanah and Weinland, 1990). To effectively manage forests for carbon, a unique set of silvicultural treatments should be tailored to the biophysical and social characteristics of each site (Ashton and Peters, 1999; Ashton et al., 2001a). Managing stands within the broader context of the landscape allows land managers to identify zones of high carbon value and stands of riparian, wetland and watershed value, as well as areas of high biodiversity. This landscape scale template should reflect an integrated network of stands allocated to production and protection (Ashton, 2003) with the focus on maximizing carbon storage within the landscape.

### *Strategic harvest planning*

One of the first steps that must be taken to manage a tropical forest for carbon is to develop a long-term strategic harvest plan. Unlike forest management in the temperate developed world, tropical forests are often managed ad hoc without long term planning (Uhl et al., 1997). Tenuous ownership rights, abundance of timber resources, and high demand all come together to provide a disincentive to take a long-term forest management perspective (Uhl and Kauffman 1990). It is important to develop a strategic harvest plan that answers the following question: What type of harvesting must be done and what type of treatments can or should be applied to retain and increase carbon over the long-term? This is vital for providing the foundation necessary to manage tropical forests for carbon. In a sense, this is indirectly related to land tenure and long term security in making management decisions now that will only bear fruit many years hence. Short term concessions of 10-60 years obviously counter such strategic planning yet they make up the majority of land tenure arrangements on government forest reserves in the tropics.

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The use of stands as the management unit within tropical forests has been largely disregarded in favor of large scale and broadly applied management prescriptions.

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### *Changing rotational length*

A key component of managing tropical forests for maximum carbon storage is the length of harvest rotations (the return time between harvests). The most frequently prescribed logging cycle in tropical forests is 30 years (Sist et al. 2003), but cutting cycles of 60 to 100 years have been found more likely to sustain timber yields and allow for increased carbon storage (Dykstra and Heinrich, 1996; Ashton et al., 2001b, Sist et al. 2003; Ashton, 2003). Studies in Brazil by Sist and Ferreira (2007) that focused on timber volume reported that after harvesting 21 m<sup>3</sup>/ha from a moist lowland forest, the next planned harvest, 30 years later, would yield only 50% of the first harvest. Despite this study's focus on timber, the results can be extrapolated to biomass and carbon storage.

On a whole, very little research has been conducted on the effect of harvesting cycles on carbon storage in the tropics. Similar studies based within the temperate regions (Cooper, 1983) have found that stands managed for maximum sustained yield store approximately a third of the carbon stored in unmanaged late stage forests. Given the increased growth rates and biomass levels of tropical forests and the results

based on timber yields, it is reasonable to expect that increased rotation lengths will increase carbon storage in tropical forests.

## **SILVICULTURAL TREATMENTS FOR MANAGING TROPICAL FORESTS FOR CARBON**

Silvicultural treatments can often be applied within tropical forests to maximize carbon storage, but the degree and intensity with which these interventions need to be applied varies. Where the harvested species are represented by abundant advanced regeneration, RIL alone could be sufficient to sustain long-term carbon sequestration and timber yields as long as logging intensities are modest and cutting cycles are long (de Madron and Forni, 1997; Sist et al., 2003; Sist and Ferreira, 2007; Valle et al., 2007). In other instances, various intensities of silvicultural treatments can be applied both before and after logging operations to promote increased carbon storage as well as to improve the overall health and productivity of the forest. The following section outlines different silvicultural methods and what each one's effect is on forest carbon.

### **Stand level planning: Regeneration**

#### *Species management based on carbon*

The logging of large commercial trees from primary forest results in disproportionate reductions of carbon stocks. This is because tree species that contribute the most to forest carbon storage are often the highly desired commercial timber species (Kirby and Potvin, 2007). The selective cutting of these high-carbon timber species dramatically reduces carbon storage within a forest and multiple decades must pass before it is gained back. These findings indicate that efforts to improve carbon storage need to be based on management techniques that promote and encourage long-term regeneration of high-carbon (late successional) species.

Pre-harvest planning combined with species selection allows forest managers to prioritize species, using as criteria (i) the species' overall contribution to carbon storage in the landscape; (ii) their relative abundance; and (iii) their per capita contributions to carbon storage (Kirby and Potvin, 2007). These steps allow forest managers to assess overall forest carbon storage and decide whether silvicultural treatments should be applied and, if so, what treatments will increase carbon storage.

#### *Site preparation*

This is the term applied to the treatments used in order to make a site suitable for either natural or planted regeneration by reducing and controlling competition. Many of the treatments are applied to improve establishment and growth of the desired species (Smith et al., 1997). In other cases, treatments are not intended for regeneration, but instead are intended for site protection as well as restoration of productivity and floristics (Smith et al., 1997). When managing tropical rainforest for carbon, preparation treatments that could affect, expose or reduce soil carbon should be minimized and treatments that free growing space for desired species should be encouraged. Liana (vine) cutting is an example of this type of treatment. After a regenerative disturbance, fast growing pioneer species can occupy the growing space

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Very little research has been conducted on the effect of harvesting cycles on carbon storage in the tropics.

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rapidly. It is at this stage that lianas can establish in a site and become harmful competitors of the desired tree species (Ashton et al., 2001a,b).

Research has shown that active measures to eliminate lianas need to be taken from the beginning of the establishment cycle. The best way of controlling them is to avoid any disturbance to the mineral soil and to maintain the site's growing space fully stocked. Liana removal affects carbon storage because it increases the light available to trees and reduces competition, allowing growth rates and carbon to increase in the stand (Wadsworth and Zweede, 2006; Keller et al., 2007; Zarin et al., 2007). It is important to notice that the positive benefits of liana removal persist only for about four years, requiring repeated treatments over a cutting cycle (Pena-Claros et al., 2008a,b).

On the other hand, treatments like prescribed burning and scarification (e.g., exposure of mineral soil) may be necessary to encourage regeneration when managing dry seasonal tropical forests for carbon. Both these treatments could have, to some degree, a negative effect on carbon storage if done inappropriately because they could reduce soil organic matter. When considering fire, the main goals are frequent fuel reduction and control of competing vegetation (Smith et al., 1997). On their own, these objectives seem counterintuitive for carbon storage and uptake; however, fuel reduction increases forest resilience to more catastrophic fires and competition control will allow the desired regeneration to take hold on the site and occupy the growing space faster. From this perspective, both treatments will have a positive effect on long-term forest carbon storage and uptake.

### *Reproduction methods*

The main goal of these treatments is to maintain ecosystem structure and function while allowing the regeneration of desirable species (Montagnini and Jordan, 2005; Smith et al., 1997). When managing tropical forests for carbon, managers should seek to increase forest structure and guild diversity and, by doing so, overall forest resiliency. A resilient forest will preserve the carbon stock in existing vegetation and actively absorb carbon in new vegetation through regeneration. The regeneration method most likely to be used to achieve these goals is the shelterwood and its variations around structural retention and age class (Smith et al., 1997; Ashton et al., 2001a,b). This method will result in initial carbon loss because of the amount of volume that needs to be removed in order to free growing space to establish the new cohort. However, a forest that is rich in species and has a diverse vertical stratification will be more resilient to catastrophic disturbances, like fires and insect outbreaks, and therefore to carbon loss in the long-term. This tradeoff between some carbon loss (to secure regeneration) and the potential total loss that could result from having the site burn down is the type of decision forest managers need to weigh when managing forests for carbon.

### *Enrichment planting*

Enrichment planting is also known as line, gap, strip or under-planting, depending on the nature of the planting arrangement (Montagnini and Jordan, 2005; Smith et al., 1997). Enrichment planting is a method utilized to introduce desirable tree species in degraded forests or stands without affecting the structure or composition already

present in the site. The introduced species can differ in their rate of growth, shade tolerance, ecological characteristics, and economic value. Choosing one species over another needs to be done paying careful attention to the issue or desired value that is being addressed (Montagnini and Jordan, 2005; Ashton et al., 2001a; Ashton 2003).

When managing forests for carbon, enrichment planting has the potential to maximize carbon uptake and storage, maintain forest structure and stratification, and increase economic value by also introducing non-timber forest products (Ashton et al. 2001b; Ashton 2003). Together, these added values could prevent the conversion of forests into other widespread, low carbon land uses (Montagnini and Jordan, 2005).

### **Non-timber forest products complement timber production**

*A case study in Sri Lanka suggests that rainforest can be managed sustainably. Herbaceous shrubs (i.e. Cardamomum zeylanicum – cardamom) can be judiciously cultivated around advance regeneration of the new forest immediately following timber harvesting. In some instances, medicinal vine species such as Coscinium fenestratum or climbing palms like C. zeylanicus (rattan) can be line-planted along edges and trails and then harvested for their economic value. Caryota urens (Fish tail palm) can be under-planted within the regenerating stand and later tapped for sugar when it matures as a subcanopy tree. The logic behind these plantings is that if lianas and other shrubs and trees grow compatibly with the timber trees, the best option would be to promote and then sequentially harvest them over the time the timber trees attain maturity. Together with services values for carbon and water, maintaining and managing a tropical rain forest for a diversity of products create greater economic value than land clearance and cultivation for tea, for example. However, timber alone cannot compare with the financial rewards of intensive tea cultivation (Ashton et al., 2001b).*

### **Stand level planning: Post establishment**

#### *Stratum treatments*

In the tropics, as in the temperate zone, the approach to managing natural mixtures of trees has often been to simplify them by converting them to single species stands or to monoculture plantations of fast-growing, sometimes exotic species. However, tropical forests are complex and stratified, and a more sophisticated understanding is necessary to maintain their composition and structure. Two stratification processes are a factor in the dynamics of species mixtures (Ashton and Peters, 1999; Ashton et al., 2001a). “Static” stratification refers to the late-successional species that occupy distinct vertical strata in the mature forest canopy (i.e. species that will always occupy understory and subcanopy positions at maturity). “Dynamic” stratification refers to the sequential occupation of the canopy strata by species of different successional status (i.e. shorter-lived canopy trees that relinquish their canopy position over time to longer-lived species) (Ashton and Peters, 1999; Ashton et al., 2001a; Ashton, 2003).

If a forest is being managed for carbon uptake and long-term carbon storage, understanding these processes of stand dynamics will provide managers with basic guidelines in the selection of appropriate regeneration methods and thinning treatments for a site. Mixtures that exhibit diverse growth patterns, and differences in shade tolerance and stand development are the best for long-term carbon storage (Ashton et al., 2001a). These stands can have shade tolerant species growing in the understory while the canopy is occupied by late-successional shade-intolerant, high carbon species. In the case of seasonal tropical forests, the lower stratum is often occupied by evergreen species that continue storing carbon even when the deciduous canopy species slow their photosynthetic activity.

Keeping track of the “book-keeping” of stratification is therefore an essential prerequisite for deciding when, where and which silvicultural treatment to use. For example, treatments can: i) accelerate shade tolerant species into the canopy strata; ii) promote shade tolerant understories to establish; and iii) allow shade intolerant canopy tree species to re-establish in the understory. All require knowledge of the current status of stand condition and stratification process to efficiently promote different aspects of tree growth and hence carbon sequestration and storage.

### *Thinning*

Thinning is directly linked to nature's self-thinning process. Self-thinning refers to the reduction in the number of trees in a stand from mortality due to continued and vigorous competition and natural selection with other individuals (Smith et al., 1997). The intent of thinning as a management practice is to purposefully regulate and manipulate the distribution of growing space at the stand level to maximize net benefits over the whole rotation before nature does this through self-thinning. Thinning therefore re-allocates growing space to remaining desired trees from competition with undesirable trees (Smith et al., 1997).

When considering the possible effects of these treatments on forest carbon, it is important to remember that the long term objective of thinning is usually to increase the size of the individual tree and/or volume of merchantable wood within a stand. This implies that the initial application of the treatment will result in a loss of standing above-ground carbon because of the reduction in the site's gross carbon volume. The amount of growing space occupied by and wood volume of the remaining trees will dramatically increase, along with a parallel increase in forest carbon (Smith et al., 1997).

This difference between merchantable wood volume yield and gross biomass production (e.g. carbon) highlights the decisions and tradeoffs between timber and carbon management that land managers will need to make when deciding which silvicultural practices to implement. The goal of many forest managers with both timber and carbon interests is to maintain site merchantable yields while obtaining some baseline long-term carbon storage. This can be accomplished by favoring allocation of growing space to highly valuable timber tree species with high carbon storage (high wood density) that are long-lived species (i.e. require longer rotations). In addition, if trees that are harvested and milled receive credit for storage as wood

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If a forest is being managed for carbon uptake and long-term carbon storage, understanding these processes of stand dynamics will provide managers with basic guidelines in the selection of appropriate regeneration methods and thinning treatments for a site.

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products, or as substitutes for more energy-intensive construction materials, promotion of thinning may appear very compatible with carbon management.

## **CONCLUSION AND MANAGEMENT AND POLICY IMPLICATIONS**

### **Summary conclusions**

- Tropical forests emit approximately 17% of total annual global greenhouse gas emissions. For this reason maximizing carbon uptake and storage while preventing loss of carbon-rich forests are important strategies currently being discussed under REDD+.
- The carbon uptake and storage capacity of a given forest varies greatly depending on the region, forest type, geophysical characteristics, species composition, disturbance regime, site degradation, land tenure, and human use.
- To develop and implement adequate forest management strategies, first it is important to understand that most tropical forests are NOT managed, but exploited.
- Implementing stand-level land use delineation, harvest planning, and reduced impact logging techniques can have important effects on increasing tropical forest carbon.
- If the goal is to maximize carbon uptake and long-term carbon storage, more complex silvicultural treatments need to be implemented. This approach will help secure the regeneration of the desired species and the continued vertical stratification of the stand, will increase productivity, and will promote the presence of target species of high economic and carbon sequestration value.
- Successful forest management can only result from tailoring silvicultural treatments to the specific requirements and characteristics of each site.
- If the silvicultural treatments described in the previous bullet points can be achieved, the forests will be more resilient to the unpredictability of disturbance and climate change, making them suitable as stable long-term carbon sequestration and storage reservoirs.

### **Implications**

#### *Areas for further investigation*

- While abundant literature exists about managing temperate forests and soils for carbon, more research is needed on how the application of silvicultural practices affects carbon uptake and storage in tropical forests at all levels.
- Future research needs to move beyond reduced impact logging (RIL) and focus on how forested landscapes can be managed for carbon, as well as water, biodiversity, and other ecological values.



### *Land managers*

- Land managers in tropical forests need to delineate stands and use them as the managing unit within the forest landscape. This would allow them to develop unique silvicultural techniques that are site specific. Stand delineation also helps identify and protect wetlands and riparian corridors, and areas of high diversity (Ashton, 2003).
- Land managers should not manage tropical forests only for timber production, but to maximize and diversify the services and products they obtain from their forests. This approach will provide an increase in net present value and a possible solution to the problem of exploitation and land conversion (Ashton et al., 2001b).

### *Policymakers*

- Policies need to focus on the preservation of standing tropical forests since almost all management and silvicultural practices applied to primary forests will result in reduced carbon storage levels.
- While it is important to implement RIL practices, it is necessary to develop policies that go beyond RIL and begin to address long-term resource needs as well as maximizing carbon uptake and storage.
- Policies that allow local and state cooperation need to be considered for managing state land. Mutual cooperation offers the possibility of maximizing the net value of the forest, therefore generating higher social, economic, and ecological sustainability (Ashton et al., 2001b).
- In comparison with data from temperate forests that indicate that some forestry practices have a minimal impact on soil carbon and this pool might not need to be measured all the time, in the tropics, data for soil carbon are lacking. For this reason it may be necessary to include all carbon pools (above- and belowground biomass, soil) when developing carbon legislation.

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## Chapter 10

# Managing Carbon Sequestration in Temperate and Boreal Forests

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### EXECUTIVE SUMMARY

In order to better understand the ways in which future forests will change and be changed by shifting climates, it is necessary to understand the underlying drivers of forest development and the ways these drivers are affected by changes in atmospheric carbon dioxide concentrations, temperature, precipitation, and nutrient levels. Successional forces lead to somewhat predictable changes in forest stands throughout the world. These changes can lead to corresponding shifts in the dynamics of carbon uptake, storage, and release.

A review of published literature on this topic revealed the following general trends:

- Drainage of wetlands for increased tree production can result in either net carbon gain or loss, depending on how deep the drainage.
- Thinning causes a reduction of the vegetative carbon pool, which recovers over a matter of decades, but the impact on soil carbon appears very limited.
- Resiliency treatments (such as fuel reduction thinning and prescribed fire) result in lowered vegetative carbon storage, but they help produce forests that are significantly less susceptible to catastrophic disturbance (with accompanying drastic carbon release).
- Regeneration harvests significantly reduce the carbon stocks in vegetation and also cause a transient increase in soil respiration, although the annual rate of carbon uptake will be greater in the regenerating stand. Harvested areas often remain net carbon sources for 10-30 years, then return to sinks.
- Carbon sequestration can be increased by extending rotations, especially if maximum biomass productivity has not yet been reached.

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- Removing harvest residues (slash) for biomass utilization, to reduce fuel levels or to prepare the site for planting will reduce carbon.
- Fertilization can increase carbon storage in vegetation and reduce soil respiration rates, but this must be balanced with the carbon released during fertilizer production.

We identify the following key points to consider for carbon storage and sequestration projects in temperate forests:

- Many forest management activities result in net carbon release and thus cannot demonstrate carbon additionality. Mechanisms should be developed to credit managers who can reduce carbon loss, not simply increase carbon gain.
- Policy makers must decide where to set baselines for carbon project accounting. Where they set the baseline determines what activities are incentivized.
- The risk of leakage must be addressed. If carbon sequestration strategies simply displace timber harvests from one forest to another, the ultimate carbon gain is questionable.
- Consideration of forest products, energy used in management operations, and forest energy substitutes determine whether or not practices like thinning are positive, neutral or negative.
- Studies have shown that many forest practices have a minimal impact on the soil carbon pool, which is the most difficult pool to measure. Thus, it may be possible that offsets involving certain forestry practices could go forward without strict quantification of this pool. This should be tempered by the fact that little is known about the affects of harvesting on deep soil carbon pools.

**Keywords:** carbon sequestration, additionality, drainage, fertilization, thinning, regeneration harvests, resiliency, rotation length, baselines, leakage

## INTRODUCTION

This chapter aims to answer the central question of how temperate and boreal forests can be managed so as to sequester carbon and contribute to climate change mitigation. Forests play a major role in the mitigation of climate change, primarily through their ability to assimilate carbon dioxide and sequester it in their living tissue, and in their long-term contribution to soil carbon stocks. Forest systems cover more than 4.1 billion hectares, or one third of the Earth's land area (Dale et.al., 2001), and temperate and boreal forests make up roughly 49% of this total.

Whether these forests are a sink or source of terrestrial carbon depends on the balance of processes that cause carbon sequestration (i.e. photosynthesis, peat

formation) and release (i.e. increased respiration, forest disturbance). Taken as a whole, the temperate and boreal forest biomes were carbon sinks during the 1980s and 1990s (Schimel et. al., 2001), but this may no longer be the case because the Canadian boreal forests are poised to release massive amounts of carbon as the result of die-off from insect infestations (Kurz, 2008). The temperate forest sink has been consistently growing with the abandonment of marginal agricultural lands, and does not experience the same scale of disturbance-mediated carbon release as in the boreal zone.

The role of forests, specifically boreal and temperate forests, in climate change mitigation has already stimulated an enormous amount of dialogue. Some of it is based on sound silvicultural practice, but much is based in myth and misconception. It is the authors' intent to address these misconceptions in an effort to align the discourse with basic principles of forest biology and silvicultural practice.

The emphasis on silviculture basics is appropriate in boreal and temperate forests because increasing forest carbon stocks in this region is a matter of making adjustments to existing forests and not undergoing a radical change in land use. Most boreal and temperate forests are second growth (Whitney, 1996) and land conversion is minimal when compared to other regions of the world. Therefore, providing additional carbon storage is a matter of refining silvicultural practices to take advantage of site nuances and enhancements.

## BOREAL AND TEMPERATE FORESTS OF THE WORLD

Boreal forests comprise the northernmost forest biome of the world, covering much of Alaska, Canada, Fennoscandia, Russia, northern Mongolia and northeast China. Boreal forests are characterized by simple, often single layered stand structure, low tree species diversity (only six genera dominate the entire range: spruce (*Picea*), fir (*Abies*), pine (*Pinus*), larch (*Larix*), birch (*Betula*) and aspen (*Populus*) and well-developed bryophyte (moss and lichen) communities. Organic-rich peat soils in boreal forests and bogs are the largest carbon pool in the biome.

Boreal forests can be roughly divided into two major zones – interior continental and maritime (Figure 1). As the name implies, interior continental forests are exposed to cold, dry continental climates. Fire and large-scale insect outbreaks are the dominant disturbance agents. In North America, interior continental boreal forests are dominated by white spruce (*Picea glauca*), Jack pine (*Pinus banksiana*), and spruce-aspen (*Picea-Populus tremuloides*) mixedwoods. In Eurasia, interior continental forests are found east of the Ural Mountains. Siberian larch (*Larix sibirica*) and *L. gmelinii*, adapted to extreme cold, drought, and permafrost, cover much of this area.

Maritime boreal forests are found in North America along the Pacific coast (Cordillarean type) and Atlantic coast (Maritime type). In this moderated climate, fir (*Abies*) species compose a larger proportion of forest area, and fire gives way to insect outbreaks and industrial forest management as the primary disturbance agents. Maritime forests are also found in Fennoscandia and northwest Russia near the Norwegian, Baltic and White Seas. Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) are the canopy dominants, with a considerable component of

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The role of forests, specifically boreal and temperate forests, in climate change mitigation has already stimulated an enormous amount of dialogue. Some of it is based on sound silvicultural practice, but much is based in myth and misconception.

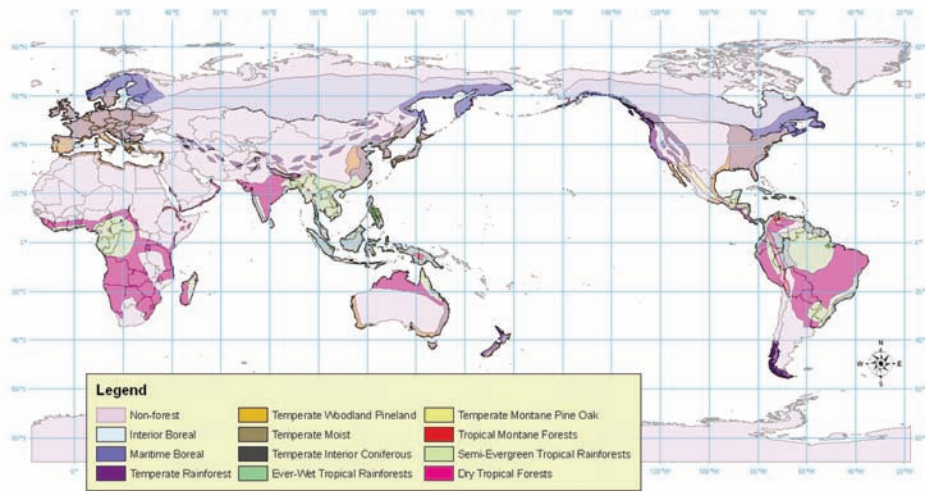
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aspen and birch. Ground fires, insect outbreaks, and industrial forestry are major influences.

Temperate forests include a wide range of forest types, and the exact boundaries with boreal forests to the north and tropical forests to the south are not always clear. A much greater range of conditions and species are present than in the boreal. Generally speaking, the soil carbon pool does not play as large a role here, while the prominence of the vegetative pools increases.

**Figure 1** Original extent of boreal, temperate, and tropical forest types of the world prior to land clearing



We describe five major temperate forest types:

1. *Moist broadleaf and coniferous forests*: mesic, mixed forests with a rich suite of genera, including maple (*Acer*), oak (*Quercus*), birch (*Betula*), beech (*Fagus*), ash (*Fraxinus*), poplar, aspen (*Populus*), hemlock (*Tsuga*), and “soft pine” species of the genera *Pinus*, spruce (*Picea*) and fir (*Abies*). Fire plays a relatively minor role in such forests except for the “hard pine” dominated forests of sandy coastal plains such as the U.S. south. They are located in the eastern United States and Canada, northern and central Europe, and the Russian Far East. Soils classified as ultisols (USDA, 1975) underlie much of this area, particularly in North America, and are generally desirable for cultivation because they are usually relatively fertile (though often stony) and require no irrigation because of precipitation year round.
2. *Interior coniferous forests*: dry, fire-adapted forests in harsh continental climates, often with andisol soils. “Hard pines” (*pinus*), spruce, fir and larch predominate. Located in the interior west of the USA and Canada, and in Central Asia, these forest types are closely related to interior continental

boreal forests. Soils are young, rocky, often skeletal, and exposed to the extremes of cold winters and dry summers.

3. *Montane oak/pine forests*: *Pinus*- and *Quercus*-dominated systems in mountain ranges of Mexico and Central America, the Himalayas, the Mediterranean and Turkey. They are fire-adapted and relatively dry.
4. *Woodland and pineland forests*: Fire-adapted, often open forests in dry, southern climates. They include “hard” pine and oak in the coastal Mediterranean region, *Acacia-Eucalyptus* savannas of Africa and Australia, and oak-pine woodlands of México. Soils that are generally classified as *altisols* (USDA, 1975) predominate. Such soils are more fertile than *ultisols* but often require partial irrigation because of drier summers. Most forests with *altisols* have already been cleared for cultivation, thus this type is restricted to degraded relics.
5. *Temperate rainforests*: Mesic, constantly moist, and often extremely productive forests of mountain ranges along coasts. Spruce, hemlock, Douglas fir (*Pseudotsuga*) and western cedar (*Thuja*) dominate in the Pacific Northwest, the southern beech (*Nothofagus*) in Chile, and southern beech, Eucalypts (*Eucalyptus*) and podocarps (*Podocarpus*) in New Zealand and Australia. *Spodosols* and *andisols* are the predominant soil types. *Andisols* are volcanic soils that with high precipitation can be very productive for pasture. *Spodosols* are acidic soils associated with bedrock geology that predominantly comprise minerals such as quartz and silica, and are therefore often nutrient poor.

## KEY CONCEPTS

Before describing the carbon impact of specific forest management actions, we would like to introduce a few key concepts and dichotomies that surface frequently in the scientific debate surrounding forest carbon storage. These concepts pertain to basic biological dynamics of carbon uptake and storage, and also to important differences in how we quantify carbon pools in managed forests.

### Maximizing carbon uptake vs. maximizing carbon storage

Biomass productivity is maximized relatively early in development, at the year when annual growth increment dips below the average annual growth increment over the age of the tree or stand. After this point growth slows, and carbon uptake slows along with it. However, while older trees (and stands) may demonstrate reduced uptake rates, the cumulative carbon stored within them can greatly exceed that of their younger counterparts. Greater pools of soil and litter carbon in older forests may also contribute to this effect, although their pattern is less clear than that of the vegetative pool.

The importance of this dichotomy lies in its management consequences. Managing for productive young forests promotes maximal carbon uptake, while

maintaining old forests and extending rotations leads to larger on-the-ground carbon stocks. In theory, a series of short rotations can sometimes lead to greater total carbon storage than a single long rotation because the stand is growing at a rapid rate for a greater proportion of the time. But each harvest entry is also followed by a release of carbon associated with increased decomposition.

#### Maximizing C uptake vs. maximizing C storage




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Managing for productive young forests promotes maximal carbon uptake, while maintaining old forests and extending rotations leads to larger on-the-ground carbon stocks.

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*These two images demonstrate the contrasting strategies of growing vigorous young forests with high rates of carbon uptake (left), and growing forests to older age classes at which uptake rate is lower, but actual quantities of stored carbon are greater (right). The downward pointing arrows indicate carbon uptake through photosynthesis, the rates of which are indicated by arrow size. Upward arrows indicate C release through auto- and heterotrophic soil respiration. In the old forest shown on the right, the inputs and outputs are near equilibrium, while on the left, uptake clearly exceeds carbon loss. However, note that the actual size of the aboveground biomass, litter and belowground biomass are considerably larger in the older forest. Importantly, the size of the soil pool does not differ much between the two examples.*

#### Site and climatic factors limit the carbon storage potential of vegetation

In any given forested site, the maximum potential productivity and carbon storage of vegetation is determined by soil fertility, moisture conditions, and climate. These factors can be regarded as placing a “ceiling” on biomass production. Forest managers can manipulate and re-allocate that biomass in different arrangements of vegetation. But to create additional carbon storage requires addressing the basic productivity constraints, for instance by fertilizing, irrigating or draining the site. As will be discussed, each of these actions carries its own carbon consequences, especially for the soil carbon pool.

A major caveat, however, is that forests may not reach their “biomass ceiling” until a quite advanced age, often much longer than the rotations used in forest management. Thus, it is often possible to create carbon additionality simply by growing forests on longer rotations so that they have time to accumulate higher standing volumes.

### **The carbon impact of an activity changes if we include the forest products carbon pool**

To illustrate, consider the example of thinning. This practice results in a reduction of the vegetative carbon pool. It is possible that the residual trees will eventually replace the biomass lost, and the pool will equal its pre-treatment size. But due to the productivity constraints described above, the pool can never *exceed* pre-treatment conditions. This makes thinning a carbon-negative or at best carbon-neutral activity *unless* we consider the sequestration of carbon within forest products (that is, we consider products to be another “pool”). If we do, thinning can appear carbon-positive, because some portion of the harvested carbon may be trapped in long-term forest products, while the residual trees are growing at a faster rate and taking up more carbon.

A great deal of literature exists on the topic of whether a forest products pool should be considered in carbon accounting, and how to quantify it. It is beyond the scope of this chapter to address this question, but it is important to recognize the impact it has on perceptions of the carbon impact of forest practices. We direct the readers to Chapters 12 and 13 in this volume for a comprehensive discussion of the forest products pool.

### **Resiliency: maximum carbon storage at high risk vs. reduced carbon stocks at reduced risk**

Forest managers have long recognized that maximizing the density of biomass on a site can be detrimental to forest health. Density-related competition often results in spindly, poorly-formed trees that are not windfirm, are susceptible to insect outbreak, and pose a fire risk. On a larger scale, the risk of such disturbances is also increased when a large proportion of the landscape is maintained in high-density, maturing stands within a limited age class range. Foresters address these concerns by managing for stand- and landscape-level resiliency. Stands are often managed at lower than maximum densities, in order to reduce risk of catastrophic loss. A sacrifice in biomass is made in order to produce fewer, larger, more vigorous trees. Across the landscape, a diversity of age classes and species is maintained.

This principle still applies when carbon uptake and storage is the management goal. Carbon stored in fire-, insect- or windthrow-prone trees and stands is “risky,” and some sacrifice in total storage may be necessary to ensure that sequestration is long-term.

### **Creating carbon additionality vs. minimizing carbon loss**

Because of the structure of many carbon credit and offset systems, the primary goal of managing forest carbon is often to create additionality. Certain practices are regarded as reliably “additional,” such as afforestation (unless by changing the site a large soil carbon loss is incurred). However, the manipulation of standing forests more commonly results in reductions of carbon pools. Such practices can be adapted in certain ways to reduce their negative carbon impact, such as by leaving more harvest residues or causing less damage to residual trees during harvest. This can result in a form of additionality, compared to business-as-usual management

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Stands are often managed at lower than maximum densities, in order to reduce risk of catastrophic loss. A sacrifice in biomass is made in order to produce fewer, larger, more vigorous trees.

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techniques. Activities such as reduced deforestation and reduced impact logging appear additional when compared to such a business-as-usual baseline.

## CARBON IMPACTS OF SPECIFIC FOREST MANAGEMENT PRACTICES

### Afforestation and reforestation

We will now give a brief introduction to the topic of reforestation/afforestation as it relates to the temperate and boreal forests (more detail can be found in Chapter 11). Afforestation and reforestation are silvicultural treatments that can often demonstrate carbon additionality. For example, the average net flux of carbon attributable to land-use change and management in the temperate forests of North America and Europe decreased from a source of 0.06 PgC per yr during the 1980s to a sink of 0.02 PgC per yr during the 1990s (Houghton, 2003). In the United States this carbon sink is overwhelmingly due to afforestation /reforestation rather than active management or site manipulation (Caspersen et al., 2000). Even though some studies suggest that as forests age the strength of the carbon sink is reduced (and may become a source under certain circumstances), the amount of carbon stored on a forested site is significantly more than any other land use (Vesterdal et al., 2007).

Land conversion to forests is typically driven by wood demand and not carbon sequestration and it is unlikely that this will change even as carbon markets develop (Eggers et al., 2008). The conversion of land to forests using passive “natural” regeneration has been postulated as an option for carbon sequestration because of the low operating costs and potential for co-benefits such as habitat formation and water quality enhancement (Fensham and Guymmer, 2009). These co-benefits provide valued ecosystem services, but make proving that the intent of the project was strictly for carbon sequestration (additionality) complicated. Rules for proving additionality are unclear and uncertain, so landowners hoping to invest in afforestation/reforestation need to make clear that the intent of the project was to sequester carbon.

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Land conversion to forests is typically driven by wood demand and not carbon sequestration and it is unlikely that this will change even as carbon markets develop.

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### Drainage

Drainage is implemented where excessive soil moisture stunts or prohibits the growth of trees. Within the boreal and temperate zones, this practice is most prominent in Fennoscandia, particularly in Finland. Drained peatland forests constitute 18-22% of the total managed area of that country (Minkinen et al., 2001). Afforestation of drained peatlands has also occurred on a large scale in Great Britain and the coastal mires of the southern United States. These peatlands areas are associated with high levels of soil carbon storage, but also with emissions of CH<sub>4</sub> (methane), an important greenhouse gas.

The carbon consequences of this drainage depend on whether the factors that increase sequestration (increased vegetative production, increased litter input, and decreased methane release) exceed the increased respiration caused by oxidation of previously anoxic peat. A critical factor in this balance appears to be how much the water table is lowered in the drainage process. When the water table was lowered from 0-10 cm to 40-60 cm (below the surface) in Finnish mires, CO<sub>2</sub> loss increased 2-3

times and stayed at that rate for at least three years (Silvola, 1986; Silvola et al., 1996). At this rate, Silvola (1986) found that such mires would switch from a modest carbon sink to a strong carbon source. Similarly, Cannell et al. (1993) hypothesized that deep drainage of peaty moorlands in Britain for Sitka spruce (*Picea sitchensis*) afforestation would result in sufficient drying such that all but the recalcitrant peat component would decompose. They concluded that carbon sequestered in the planted trees, increased litter layer, and wood products would not make up for this loss.

In contrast, when the water table in a Finnish mire was only lowered 5-9 cm, emissions barely changed (Silvola et al., 1996). Similarly, afforestation of Irish moorlands did not result in deep drying and oxidation. Increased CO<sub>2</sub> release was minimal, and exceeded by increased storage in biomass and litter. In fact, there was little increase in efflux even from well-drained peats (water table reduced to 50 cm below surface), suggesting that the increased microbial activity that causes respiration might be limited to upper layers by some other factor than moisture (Byrne and Farrell, 2005). Von Arnold et al. (2005) examined CO<sub>2</sub> and CH<sub>4</sub> efflux (which are usually negatively correlated) in undrained, lightly drained and well-drained (dry) peatlands in Sweden. They found that, from the perspective of minimizing greenhouse gas emissions, the optimal condition was lightly drained peat, because increases in CO<sub>2</sub> efflux were exceeded by the decrease in CH<sub>4</sub> efflux. In contrast, both undrained and dry peats were carbon sources to the atmosphere. Importantly, this analysis did not consider the additional sequestration potential of enhanced tree growth and litter production.

When the biomass and litter pools are considered, even greater carbon gains have been recorded in Sweden, Finland and Russian Karelia (Laine and Vasander, 1991, Minkinen and Laine, 1998, Sakovets and Germanova, 1992). Drained, plowed and afforested peatlands in Scotland were a carbon source for only 4-8 years, at which point increased vegetative productivity switched them to sinks. This effect only increased as the forests matured (Hargreaves et al., 2003).

Thus, drainage of peatlands for increased forest productivity has the potential to be carbon positive or carbon negative, depending on how thorough the drainage is. Shallowly drained sites tend to sequester more carbon than undrained sites because increased tree growth and decreased methane emissions outweigh increased CO<sub>2</sub> emissions. The opposite is true on deeply drained sites.

### **Fertilization**

Tree growth in temperate regions is typically nitrogen-limited. Therefore, nitrogen fertilization is a well-established treatment in this region to increase biomass production. This increased capacity to store carbon is well documented, but must be considered in light of the carbon emissions required to produce and apply the fertilization treatment.

Biomass production is the result of the energy produced by photosynthesis, minus the respiration requirements of the non-photosynthetic plant tissues. Higher fertility increases leaf area, nutrient concentration, and carbon assimilation rates and in turn, improves carbon availability and overall biomass production (Coyle and Coleman,

2005). Nitrogen fertilization has been shown to increase biomass production as much as 16 Mg per ha over 100 years in some intensively managed pine forests in the southeastern United States (Markewitz et al., 2002). On some low fertility sites, nitrogen fertilization can make the difference between the site's being a carbon source or a carbon sink and can lessen the time it takes for a developing stand to go from a source to a sink. The degree of effect that fertilization has depends on the baseline fertility of the site (Maier and Kress, 2000).

The fertility of a site can be approximated by determining the nitrogen-use efficiency, a measure of the amount of additional carbon assimilated as a result of the addition of a kg of nitrogen. Nitrogen-use efficiency for carbon sequestration in trees strongly depends on soil nitrogen status as measured by the carbon/nitrogen ratio. Excessive fertilization or appropriate fertilization plus the deposition of anthropogenically elevated levels of atmospheric nitrogen can cause deposition rates to exceed the capacity for nitrogen uptake, and nutrient imbalances can lead to forest decline due to nitrogen saturation (Bauer et al., 2004). The effect of nitrogen saturation is also seen in soils when the biotic component of soil is no longer able to uptake and stabilize the nitrogen in organic compounds. The excess nitrogen is leached out of the soil in the form of nitrates.

It has been thought that fertilization decreases soil carbon stocks through an increase in decomposition. However, many recent studies have demonstrated that fertilization may increase carbon stocks in the soil. Hagedorn et al. (2001) found that soil organic carbon (SOC) sequestration in fertilized plots was always higher than that in control plots. They and others conclude that fertilization of temperate and boreal forests has high potential to reduce both heterotrophic and autotrophic soil respiration (Preglitzer et al., 2008). Decomposition is slowed as a result of several factors: i) decreased carbon allocation to mycorrhizae; ii) direct suppression of soil enzymes responsible for litter degradation; iii) decreased litter quality; and iv) decreased growth rates of decomposers. The research highlighting the sequestration of SOC as a result of fertilization is relatively recent and the hypotheses about the mechanisms that drive it are primarily speculation. More research is needed to address this knowledge gap.

Similarly to nitrogen fertilization, temperature can influence soil carbon stocks in the temperate and boreal regions. Temperature can influence nutrient availability and therefore fertility. In the future, therefore, the effect of nitrogen fertilization on soil carbon storage may be offset by the opposite effect of climate change; small increases in temperature will increase the rates of decomposition and nitrogen cycling and the carbon stock of forests may decline due to accelerated decomposition of SOC (Makipaa et al., 1999). This is likely to be a gradual change, but will be most pronounced in the boreal regions where processes are typically more limited by temperature than in temperate regions.

Although nitrogen is limiting in many forests of the temperate and boreal regions, it is not the only fertilization treatment used. In nitrogen-rich sites such as drained peatlands in central Finland or poorly drained loam and clay soils of the upper coastal plain of Georgia, USA, treatments such as additional phosphorus,

calcium, potassium or liming are needed to amend critical nutrient levels or pH (Hytönen, 1998; Moorhead, 1998). In northeastern Oregon and in central Washington where nitrogen is considered limiting, research has shown that the addition of nitrogen and sulfur to Douglas-fir stands produced significant growth response to the nitrogen + sulfur treatment, but not to the nitrogen-alone treatment (Garrison et al., 2000). Similarly, in loblolly pine stands in the coastal plains of Georgia, USA, phosphorus is needed to enhance uptake of nitrogen (Will et al., 2006). Finally, in northwestern Ontario, Canada, the best treatment in terms of total volume increment over that of the control was 151 kg nitrogen per ha plus 62 kg magnesium per ha, which produced about 16 m<sup>3</sup> per ha of extra wood over 10 years (Morrison and Foster, 1995).

These examples illustrate the complexities often associated with the correct application of fertilization and amelioration treatments to increase carbon on forested sites. These treatments are site specific; a manager's mastery of the intricacies of the site is essential to increasing the carbon uptake on a site.

It is beyond the scope of this chapter to provide a comprehensive look at the trade-offs between an increase in carbon storage in temperate and boreal forests and the fossil fuel emissions that result from the acquisition, manufacture, transport, and application of fertilizers. Most results indicate that even on the sites where fertilization is most beneficial, the emissions of CO<sub>2</sub> outweigh the carbon sequestered as a result of increased biomass production and SOC stocks (Schlesinger, 2000; Markewitz, 2006). However, on nitrogen-poor sites, where appropriate, the encouragement of the establishment of nitrogen-fixing plants may be beneficial through natural or artificial seeding (Marshall, 2000).

### Thinning

Thinning is a silvicultural practice that lowers stand density through the removal of a portion of the standing volume, often at regular spacing. It clearly impacts the aboveground carbon pool, and it also affects the litter pool (through the addition of slash and reduction of post-thinning litterfall), and potentially the soil pool (through increased respiration due to increased light and warmth at the soil layer).

If thinning were a continuous process, it would have little effect on mean biomass storage. If trees could be removed singly as they succumb to mortality, their neighbors could quickly occupy the available growing space, thereby making up for the lost biomass. However, this is operationally and commercially impractical, so thinning is done periodically (Cooper, 1983). As a result, a greater amount of growing space is left unoccupied for a longer time, resulting in reduced stand volume and carbon storage. In a Ponderosa pine (*Pinus ponderosa*) stand in California, thinning reduced net ecosystem production (NEP) by a third, and pre-treatment levels were not reached for 16 years (Campbell et al., 2009). One Australian thinning modeling study suggested that it would take approximately 60 years for a thinned stand to return to the pre-treatment carbon storage level (Spring et al., 2005).

Importantly, the decrease in stand production does not always scale perfectly with the reduction in stand density. Light-use efficiency of Ponderosa pine was almost



60% higher in thinned than unthinned stands (Campbell et al., 2009), perhaps because the trees removed in the treatment were of low vigor and were not using site resources efficiently. Also, if canopy thinning stimulates increased growth in midstory and understory vegetation, reductions in aboveground net primary production can be quickly offset (e.g. thinning in Ohio oak-maple (*Quercus-Acer*) stands, Chiang et al., 2008). However, after thinning, a stimulated shrub layer can also result in net carbon loss if it has lower net primary productivity than the tree layer but similar respiration rates. This was observed after thinning of Ponderosa pine plantations in California (Campbell et al., 2009).

Different types and intensities of thinning have different impacts on carbon storage. Interestingly, pre-commercial thinning of northern cypress pine (*Callitris glaucophylla*) in New South Wales increased total stand carbon because all the cut trees remain on the ground (and are sequestered for some time in the litter pool) while the residuals accumulate biomass at a faster rate (McHenry et al. 2006). In Allegheny hardwoods, plots thinned from below showed no significant difference in carbon storage from unthinned plots, perhaps because of the low vigor and growth efficiency of the thinned trees. Crown-thinned plots sequestered significantly less carbon, and thinned-from-above plots even less (Hoover and Stout, 2007). Increasing the thinning intensity in radiata pine (*Pinus radiata*) and maritime pine (*P. pinaster*) plantations in Spain resulted in a 9-12% reduction in carbon storage (Balboa-Murias et al., 2006). In a Norway spruce plantation it led to reduced carbon storage and sequestration rate that was still evident 33 years later (Nilsen and Stand, 2008).

Thinning influences litter and soil carbon as well. In a review of forest management effects on these pools, Jandl et al. (2007) found that forest floor carbon declined with increasing thinning intensity in field studies in New Zealand, Denmark, and the USA. Litterfall additions to the forest floor and higher ground temperatures stimulated decomposition. However, the impact was moderated by the addition of logging slash to the litter layer, and the fairly rapid return to pre-treatment temperatures in all but the most intensively-thinned plots (Jandl et al., 2007). Increases in CO<sub>2</sub> efflux after thinning have been observed for several years in California mixed conifers and Ozark oak-hickory (*Quercus-Carya*) stands (Concilio et al., 2005).

The soil pool appears even more buffered from the effects of thinning than the litter pool. Some increase in soil respiration was observed after thinning in Norway spruce, but no significant effects on soil carbon storage could be detected with increasing thinning intensity (Nilsen and Strand, 2008). Thinning in South Korean forests of Japanese red pine (*Pinus densiflora*) and German European beech (*Fagus sylvaticus*) produced no significant increases in respiration (Dannenmann et al., 2007; Kim et al., 2009). In loblolly pine (*Pinus taeda*) plantations in Virginia, the contribution of logging slash and decaying roots to the soil actually *increased* soil carbon concentration in the 10-40 cm depth 14 years after thinning (Selig, 2008).

Thinning thus produces a short term decrease in vegetative and litter carbon pools, and little to no increase in soil respiration. How long this negative carbon impact lasts depends on the intensity and type of thinning, and on how fast residual trees can replace the biomass removed. Whether slash inputs to the litter layer exceed

reductions in litterfall also plays a small part in defining when pre-treatment carbon levels are re-attained.

### **Thinning and the carbon balance of a forest stand**

*Flux tower measurements taken in a 40-year-old Scots pine (*Pinus sylvestris*) stand in southern Finland showed that CO<sub>2</sub> flux did not change after the first commercial thinning. A complex of factors allowed this. A reduction in overstory photosynthesis was balanced by an increase in understory photosynthesis. And while heterotrophic respiration increased with the decomposition of logging slash and roots, this in turn was balanced by a reduction in autotrophic root respiration.*

*Thus, the “redistribution of sources and sinks is comprehensively able to compensate for the lower foliage area” in the thinned stand.*

From Suni et al., 2003

### **Resiliency treatments**

Disturbance plays a vital role in the natural flow of carbon between pools, but as a result of past management practices and a changing climate, many forests in the boreal and temperate regions have become especially susceptible to catastrophic disturbances (Hurteau and North, 2009) that release excessive amounts of carbon into the atmosphere.

Managing for carbon should strive to maximize the amount of stored carbon while minimizing the likelihood of catastrophic disturbance. This balance is achieved through maximizing forest resiliency. This section will address resiliency by first defining resiliency in terms of carbon sequestration. This will be followed by examples of management responses to disturbance such as fire, insect infestations and wind. In conclusion we will discuss management of forest resiliency for carbon sequestration in the face of changing disturbance regimes.

Here, we define resilience as the capacity of a system to absorb disturbance and reorganize while undergoing change so as to retain essentially the same function, structure, and ecosystem services (adapted from Folke et al., 2004). This definition works well for our purposes because it accounts for a resilient forest's ability to reduce carbon loss from a disturbance and reorganize in such a way that maintains high levels of the desired ecosystem service, carbon sequestration.

#### *Fire*

Fire is a dominant disturbance agent in many temperate forest regions. In some regions, uncharacteristic fire frequency and intensity is due to changing climactic conditions (Lucas, 2007). In many others, the structure of fire dependant temperate forest ecosystems has been altered as a result of a high level of fire suppression over the last 100 years (Covington et al., 1997; Allen et al., 2002; Brown et al., 2004). This

has resulted in a buildup of fuels leading to intense fires (Hessburg et al., 2005). Tilman et al. (2000) found that in an oak savannah in Minnesota, when fire was excluded, forests were able to build both above and belowground biomass to levels 90% greater than in forests with frequent ground fires. This sequestered carbon is at high risk of sudden release due to the potential for catastrophic fire. On such sites, forest managers may choose to find a balance between increased sequestration and greater assurance of long-term storage by reducing stem density and fuel loading.

The restoration of more fire-resilient forests is possible and critical (Agee and Skinner, 2005). A combination of thinning and burning can build resiliency through the removal of elevated levels of biomass from sites. Forests under such management will hold less carbon than the maximum possible, but over the long term they may hold more than forests experiencing an occasional catastrophic burn (Houghton et al., 2000). Prescribed fire treatments are intended to reduce the fuel loading without causing significant mortality to the remaining vegetation. It is well known that fire severity determines the amount of carbon released during the acute stages of the disturbance. However, some studies indicate that nearly half of the carbon released is lost through the much slower decomposition processes over a period of years (Brown et al., 2004; Hessburg et al., 2005). In fact, some experiments have shown that recently burned and harvested sites are sources of carbon, and that recovery to the same flux as a mature site can take 10 years following a fire (Amiro, 2001). Causes of this phenomenon are linked to an increase in soil respiration due to an increase in soil surface temperatures, producing a carbon source for up to 10 years. The complex interactions between fire, soils, vegetation, and site recovery from a disturbance are just beginning to be understood.

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The restoration of more fire-resilient forests is possible and critical.

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It is important to point out that there is a carbon loss associated with the use of prescribed fire. Surface soils, litter and downed woody material will be carbon sources for some years after the disturbance. Land managers need to weigh these emissions against either a no-action alternative or another silvicultural treatment to determine the best fit for the site. It should be stressed that the carbon loss from catastrophic disturbance can be extensive and long lasting and the management decision should work toward a site condition that is resilient to disturbance.

It should be kept in mind that some forest types, such as lodgepole pine (*Pinus contorta*) in the temperate zone, are adapted to catastrophic disturbance. It would thus be misguided to attempt to produce a resilient forest in the sense of one “capable of maintaining substantial live basal area after being burned by a wildfire” (Agee and Skinner, 2005). The autecology of species like lodgepole pine may make stands they dominate inherently more “risky” for carbon sequestration, and inappropriate as sites for long-term storage.

### *Wind*

Unlike fire, the magnitude of carbon loss from a wind disturbance is not so closely linked to stocking density. Wind as a disturbance agent can affect forests through a wide range of magnitude and spatial scales, from a localized downburst damaging a single tree to the large-scale damage caused by hurricanes (McNulty, 2002). The

resilience of trees and understory vegetation to wind disturbance can provide a tight biotic control of ecosystem processes like carbon sequestration, and is based on the structure of the forest prior to the disturbance (Cooper-Ellis et al., 1999). The greater the diversity of functional groups represented in the pre-disturbance forests, the greater capacity the forest has to maintain or recover the ability to sequester carbon in the environment that follows the disturbance (Busing et al., 2009).

### *Insect/Pathogen*

As the climate changes, the ability of native and non-native forest pests to establish and spread increases because the range of suitable environments expands. The door opens to insects and pathogens that previously posed less of a risk. Direct effects of climate change on forest pests will likely be increased survival rates due to warmer winter temperatures, and increased developmental rates due to warmer summer temperatures (Hunt et al., 2006). A striking example is in the interior of British Columbia where the mountain pine beetle (*Dendroctonus ponderosae*) infestation is rapidly spreading north.

#### **Managing for resiliency in forests affected by the mountain pine beetle**

*“There are literally several hundred million cubic meters of wood out there in the forests decomposing and releasing carbon dioxide back into the atmosphere,” (Kurz et al., 2008) from a massive outbreak of the mountain pine beetle (Dendroctonus ponderosae) across the lodgepole pine (Pinus contorta) forests of interior British Columbia. This infestation and subsequent catastrophic fires in beetle-killed timber are threatening to turn Canada’s forests from a carbon sink to a source. It is projected that the region could release 990 million tons of CO<sub>2</sub> – more than the entire annual emissions reported by Canada in 2005 (Kurz et al., 2008).*

*Research has demonstrated that direct management of mountain pine beetle through tree removal, burning or insecticide application is impractical and ineffective. Rather, that alteration of stand structure (age-class distribution, composition and density) has the best chance of minimizing the scale and intensity of the infestations and associated negative carbon flux from these forests (Amman and Logan, 1998). Unfortunately, because of a century long campaign of aggressive fire suppression, and an attempt to maintain a status quo of current stand conditions that goes beyond the natural cycle of regeneration and renewal, there are limited opportunities for appropriate silvicultural treatments.*

Depending upon species-specific characteristics, mixed forests may contribute to ecological stability by increasing resistance and resilience (Larson, 1995). A good example is the mixed hemlock/hardwood forests of the northeastern USA. Hemlock woolly adelgid attacks hemlock trees of all ages and sizes, and infested trees seldom recover (Nuckolls et al., 2008). Carbon effects from the infestation are not surprising; during the first year of infestation, autogenic respiration of CO<sub>2</sub> from roots is reduced

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As the climate changes, the ability of native and non-native forest pests to establish and spread increases because the range of suitable environments expands. The door opens to insects and pathogens that previously posed less of a risk.

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although no additional carbon is stored because there is little or no photosynthesis occurring. Decomposition increases as trees die as a result of increased light regimes, leading to increased soil temperatures. Overall the carbon release depends on the size of the infestation and the species mix associated with the hemlock stands. Since most hemlock stands are not single species, or single age class, the carbon loss from the ecosystem as a whole is less than in monotypic forest types such as lodgepole pine (Orwig and Foster, 1998). Additionally, large-scale stand-replacing fires are not typical in the northeast U.S. where the hemlock woolly adelgid is found. In the context of carbon sequestration, mixed hemlock/hardwood forests are more resilient to insect infestation than lodgepole pine forests because of their diversity.

### *Climate change*

If climate change alters the distribution, extent, frequency, or intensity of any of these disturbances, large impacts could be expected (Dale et al., 2001). The diversity of species in a dynamic ecosystem undergoing change appears to be critical for resilience and the generation of ecosystem services (Folke et al., 2004). In this sense, biological diversity provides insurance, flexibility, and a spreading of risk. Therefore management should attempt to strive for diverse, mixed species, multiple age class stands, or any combination thereof, for all forest types – simple or complex. It is one important tool that contributes to sustaining the response required for renewing and reorganizing desired ecosystem states after disturbance (Larsen, 1995).

Resilience can be influenced at the landscape level by the presence of refugia that escape disturbance and serve an important re-colonization function for surrounding areas. This diversity of species and heterogeneity in the landscape builds integrity, meaning that even if the disturbance causes a change in the stable state of the forest, the new stable state will function in a similar way, providing the same ecosystem services, including carbon sequestration (Perry and Amaranthus, 1997).

### **Regeneration harvests**

Regeneration harvests are silvicultural treatments that remove overstory cover to release existing regeneration or make growing space available for the germination of a new cohort. These harvests have the potential to alter the aboveground vegetation, bryophyte, litter, and soil carbon pools.

The effect on the vegetative pool depends on the type of regeneration harvest. Uneven-aged treatments such as selection harvesting may have effects similar to thinning in that they only remove a portion of the canopy cover (Laporte et al., 2003). In a comparison of harvest types in Ontario, Canada northern hardwoods, carbon storage was greater after selection harvesting than clearcutting because vigorous residual trees remained on the site (Lee et al., 2002). Clearcutting has a distinct and stronger effect. A clearcut of old-growth Norway spruce in Finland resulted in a 1/3 reduction in ecosystem carbon (Finer et al., 2003). Whole-tree harvesting on a 100-year rotation was modeled to result in an 81% reduction in biomass carbon compared to uncut forests in boreal China (Jiang et al., 2002).

Harvesting's influence on litter and soil carbon is controversial. An influential study by Covington (1981) in clearcuts at Hubbard Brook Experimental Forest in New Hampshire, introduced a paradigm of increased decomposition (and hence soil carbon loss) after forest harvest. His findings suggest that forest floor organic matter declines 50% within 20 years of harvest. A number of studies reinforce this view. In a modeling simulation of the effects of different harvest regimes on carbon stocks in boreal *Larix gmelinii* forests in China, clearcutting was predicted to result in litter and soil carbon loss that was greatest 10-20 years after harvesting, and to slowly recover thereafter (Jiang et al., 2002). A 30-year period of post-harvest soil carbon loss was observed in Nova Scotia red spruce (*Picea rubens*) forests, including from the deep mineral soil (Diochon et al., 2009).

A growing body of research, however, suggests that post-harvest respiration is not as important in the carbon budget as Covington suggested. A critical re-visit of his study suggested that the loss of organic mass from the forest floor after harvest was due to intermixing into the mineral soil, not increased decomposition (Yanai et al., 2003). If this is true, then the carbon consequences of harvesting are quite different, since organic carbon incorporated into the mineral soil may actually increase total carbon sequestration on the site.

Several comprehensive reviews of harvest effects on soil carbon also indicate limited impact. Depending on the level of slash input and organic matter incorporation into the mineral soil, harvest can result in slightly negative or slightly positive, or often no changes in soil carbon (Johnson, 1992; Johnson and Curtis, 2001). Conversion of old-growth *Picea* forests in British Columbia to young plantations reduced litter carbon stocks but left mineral soil carbon unaffected (Fredeen et al., 2007). Little or net loss of forest floor weight was associated with clearcutting or partial cutting in Canadian boreal mixedwoods, perhaps due to rapid return to pre-treatment light and moisture conditions after prolific trembling aspen (*Populus tremuloides*) sprouting (Lee et al., 2002). In both Ontario northern hardwoods (Laporte et al., 2003) and Ozark oak forests (Edwards and Ross-Todd 1983, Ponder 2005, Li et al., 2007), uneven-aged management led to increased soil carbon levels, and clearcutting to no significant change, from controls. Rates of both root respiration and microbial respiration may decline after harvest due to tree removal and soil compaction (Laporte et al., 2003). Where increased efflux has been observed, it tends to be small and limited to the uppermost soil layer (such as in a Chilean lenga (*Nothofagus pumilio*) shelterwood (Klein et al., 2008)), and recovers to pre-harvest conditions after only a few years (such as in trembling aspen clearcuts in Ontario, Canada (Weber, 1990)).

Johnson and Curtis (2001) hypothesized that whole tree harvesting could potentially result in soil carbon losses because of the high rates of biomass removal from the site. However, field studies in northern New Hampshire and Maine indicate that this practice results in no reduction in forest floor mass or soil carbon pool relative to uncut areas (Huntington and Ryan, 1990; McLaughlin and Philips, 2006). Some researchers suggest that the long-term consequences of management on soil carbon pools will be stronger than the short-term. A 300 year model of Canadian

boreal forests shows a consistent decline in soil carbon in managed forests (Seely et al., 2002). Multi-rotation monitoring of managed forests will be necessary to assess the rigor of such models.

As the above studies indicate, there is significant evidence to show that soil carbon loss after forest harvest is a short-term component of a site's carbon budget. This is based on the observation that mineral soil carbon is usually not affected by harvest, and that the loss from litter layers can be offset by slash additions. If the impact on soil carbon is indeed minor, then intensive measurement of soil carbon pools may not be necessary after traditional forest harvests. One of the main criticisms of this theory is that the research supporting it rarely involves measurement of deep soil carbon. One of the few studies to do so (in a red spruce chronosequence in Nova Scotia) found that younger post-harvest stands had significantly lower carbon storage at the 35-50 cm soil depth (Diochon et al., 2009). Before the conclusion can be made that soil carbon pools are not significantly affected by harvesting, greater attention must be paid to these deep soil layers.

If all the carbon pools, inputs and outputs are considered together, it appears that clearcut stands are carbon sources for the first decade after harvest (thanks to transient increases in respiration), after which they switch to sinks. This pattern holds for boreal forests in British Columbia (Fredeen et al., 2007), Saskatchewan (Howard et al., 2004) and Finland (Kolari et al., 2004), but its applicability in temperate zones is not as clear.

### Treatment of harvest residues

The addition of harvest residues to the litter and soil layers is an important factor in mitigating carbon loss from harvested forests. This might suggest a negative carbon influence from removing these residues (and natural litterfall) for utilization, fuel reduction or site preparation. However, research is mixed. Balboa-Murias et al. (2006) found that logging residues contained 11% of the total biomass carbon stored across a rotation in radiata pine and plantations in Spain. They thus concluded that residue harvest for biomass burning (a common practice in Spanish forests) would result in reduced ecosystem carbon storage. Piling and burning slash in California clearcuts resulted in soil carbon loss (Black and Harden 1995). Removing harvest residues alone from New Zealand radiata pine plantations did not significantly alter soil carbon levels, but removing residuals and the forest floor (i.e. accumulated litterfall) did. In addition, a pattern of increasing soil carbon stocks with increasing residue retention was observed (Jones et al., 2008). In oak forests of Missouri, there was no significant increase in soil respiration between whole-tree harvest and whole-tree harvest + forest floor removal, and both had lower respiration than the control (Ponder, 2005). In *Eucalyptus* forests in Australia, residue retention had minimal impact on soil carbon levels, but may have some influence if practiced across multiple rotations (Mendham et al., 2003).

It appears that removing logging slash from harvested sites reduces the litter carbon pool, which is important in some forest types. But unless the natural litterfall is also reduced, residue removal has limited impact on soil carbon levels.

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If all the carbon pools, inputs and outputs are considered together, it appears that clearcut stands are carbon sources for the first decade after harvest (thanks to transient increases in respiration), after which they switch to sinks.

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### Changing rotation length

Many forests in the temperate and boreal zones are managed on rotations far shorter than the potential age of the species present. Often these rotations are so short that the maximum biomass productivity possible on the site (the “ceiling”) is never reached. In a broad review of forest management effects on carbon storage, Cooper (1983) found that, on average, stands managed for maximum sustained yield store only 1/3 of the carbon stored in unmanaged, late successional forests. Managing for financially optimal rotation results in even smaller storage, perhaps 20%.

This raises the possibility of creating carbon additionality (in comparison to business-as-usual managed forests) by increasing rotation length. Much research supports the positive effect of this practice. In Chinese boreal forests, Jiang et al. (2002) modeled a variety of rotation lengths and found that 30-year rotations stored only 12% as much carbon as 200-year rotations. In Europe, rotation modeling of spruce and pine forests showed increased carbon storage with increased rotation. This is especially true where stands retain high net primary productivity (NPP) rates even at extended rotations, such as pine plantations in northern Spain (Kaipainen et al., 2004). Further research in Spain supports this finding, although the authors noted that mean annual carbon uptake eventually will decline with increasing rotation as trees become less productive (Balboa-Murias et al., 2006). Jandl et al. (2007) found that lengthening rotations would increase carbon storage until stands reached an advanced developmental stage in which biomass actually began to decline (as observed in some old-growth forests).

As is often the case, the impact of rotation length on soil carbon is complicated. One Finnish study found that soil organic matter was maximized with shorter rotations, because of increased slash inputs to the litter and soil layers (Pussinen et al., 2002). Lengthening rotations in models of wood production in Finland resulted in greater carbon storage when the increase in biomass carbon exceeded the decrease in soil organic matter. This occurred in the case of Scots pine, but not for Norway spruce, suggesting that short rotations are more carbon-positive for the latter species (Liski et al., 2001). This must be tempered, however, by the increased fossil fuel emissions associated with short-rotation forestry (Liski et al., 2001).

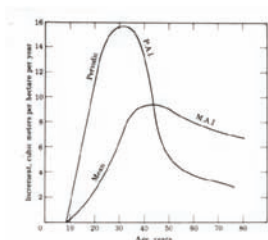
The principle behind lengthening rotations is to bring stands closer to the advanced ages at which maximum biomass is attained. By this same principle, forests that are already in these stages (for instance, old-growth) should be maintained. Harmon et al. (1990) considered the carbon consequences of the conversion of old-growth forests in the Pacific Northwest of the United States to managed production forests, finding that it caused a reduction in carbon storage that extended for 250 years, and could probably never be made up for. If forests in this region were managed with rotations of 50, 75 and 100 years, the carbon stored would be at most 38%, 44% and 51%, respectively, of that stored in old-growth (Harmon et al., 1990). Tang et al. (2009) predicted a similar long-term loss in ecosystem carbon with the conversion of Michigan northern hardwoods to younger stand structures. Managing red spruce on 60 year rotations in Nova Scotia would result in the loss of 42% of soil carbon relative to old-growth and 26% relative to 80 year rotations (Diochon et al.,



2009). Managed *Eucalyptus* forests in Australia contain only 60% of the aboveground vegetative carbon stored in old-growth.

The key explanation of this discrepancy is the dearth of large (>100 cm in diameter) trees in managed stands. In old-growth rainforest/eucalyptus stands in New South Wales, Australia, such trees make up only 18% of the stems >20 cm, but contain 54% of the vegetative carbon (Roxburgh et al., 2006). These studies suggest, at the least, that when old-growth forests already exist, their maintenance is optimal for carbon sequestration.

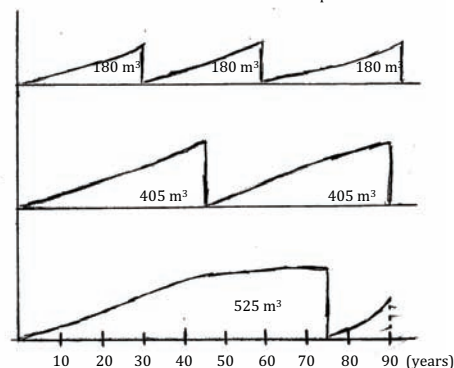
### The principle of extending rotations to sequester more carbon per hectare



The figure on the left is an example of how to set timber rotations to maximize stand productivity. When periodic annual increment (PAI) (amount of volume added per hectare in a year) equals mean annual increment (MAI) (the total stand volume divided by stand age) it is time to cut. After this point, the stand will no longer be adding as much volume per year as it has been on average across the rotation.

It is often proposed that carbon credits could be offered to incentivize landowners to extend rotations to the PAI=MAI point. In this way, more biomass will be grown on a hectare over a given time period (add the yield of each rotation) than would have been under "financial maturity" rotations. However, if rotations are extended beyond the maximum productivity point, then total biomass produced per hectare across the time period will be reduced, even though at the end of one such rotation volume/ha is at its greatest level in the example.

It should be kept in mind that this example does not consider the other carbon pools, notably the forest litter. Short rotations add more harvest slash to this pool, but there is also a period of heightened decomposition after each harvest. Research indicates that the soil pool is minimally impacted by forest harvest unless significant soil disturbance takes place.



Traditional "financial maturity" rotations of 30 years  
Total volume grown per ha across rotations = **540 m<sup>3</sup>**  
Each rotation ends before stand attains max. productivity

Peak productivity rotations (PAI=MAI) of 45 years  
Total volume grown per ha across rotations = **810 m<sup>3</sup>**  
Each rotation ends at point of max. productivity

Rotation extended beyond peak productivity (75 years)  
Total volume grown per ha across rotations = **540 m<sup>3</sup>**  
rest per rotation volume is achieved, but summed rotations produce less volume.

## MANAGEMENT AND POLICY IMPLICATIONS

Certain key themes emerged from our research regarding forest management carbon impacts. We will first summarize those themes pertinent to managers interested in sequestering carbon within their forests. We will then discuss those pertinent to forest management decision making within a climate change mitigation scheme.

- Relatively few forest practices can demonstrate true carbon additionality. Afforestation/reforestation usually increases carbon sequestration on the site, unless it results in significant release of soil carbon (i.e. through intensive site preparation or the oxidation of peat soils). The impact of afforestation/reforestation on soil carbon pools must be carefully monitored.

- Fertilization treatments that improve the nutrient conditions limiting plant growth can increase the vegetative carbon pool (particularly on marginal soils), and increase the soil carbon pool by reducing root and microbial respiration. This must be tempered by consideration of the carbon footprint of fertilizer production, which can match or exceed the additional carbon sequestration.
- Draining of saturated peat soils and subsequent afforestation can cause either a net carbon loss or gain, depending on whether increased tree growth and litterfall and decreased methane release outweigh the increase in respiration from oxidized peat. This may in turn be dependent on the extent to which drainage lowers the peatland water table. Research from drained lands in Finland and the British Isles indicates that net carbon sequestration is possible when the water table remains relatively high after drainage.
- Thinning causes reduction of the vegetative carbon pool, which recovers over a matter of decades (depending on thinning intensity and tree vigor). Thinning's impact on soil carbon appears very limited, as inputs of slash and reduced root respiration seem to make up for reduced litterfall and increased microbial respiration.
- Resiliency treatments (such as fuel reduction thinning and prescribed fire) result in lowered vegetative carbon storage and some carbon release from decomposition and combustion. However, they help produce forests that are significantly less susceptible to catastrophic disturbance (with accompanying drastic carbon release). Essentially, forest managers using these treatments accept less than maximum carbon storage to ensure longer-term and less "risky" storage.
- Regeneration harvests significantly reduce the vegetative carbon pool, especially even-aged treatments such as clearcutting. The carbon stored in this pool may not rebound for many decades (or centuries, if the pre-harvest stand was in old-growth condition), but the annual rate of carbon uptake will be greater in the regenerating stand. Harvested stands often are net sources of carbon for the first 10-30 years, because of increased litter and soil respiration. They then become net sinks as vegetative growth and litter accumulation exceed respiration.
- Removing harvest residues (slash) for biomass utilization, to reduce fuel levels or to prepare the site for planting, directly reduces the litter carbon pool. The impact on soil carbon is less clear. Treatments that only reduce slash do not result in significant soil carbon loss (over one rotation), but loss is demonstrated if the forest floor (natural litter accumulation) is removed as well.
- Managing stands for maximum sustained yield or financially optimum rotation can result in non-optimal carbon storage. Such rotations are often

too short to allow the stand to attain maximum biomass. As such, it is often possible to increase carbon sequestration by extending rotations. This is particularly true on productive sites where high rates of NPP can be sustained through longer rotations. There is a point of diminishing returns, though, when rotations are extended beyond the age of maximum biomass productivity. At some point, it may be possible to store more carbon in a series of short rotations (that maintains the stand in a young, productive stage) than a single longer rotation.

- If old forests *already* exist, however, it is almost never better to convert them to younger forests. Old-growth forests, especially in productive zones, often have very large pools of vegetative carbon in comparison to younger, managed forests. The largest trees present in old-growth forests contain a disproportionate amount of carbon, and their absence in managed forests can explain the discrepancy. Soil and litter pools may also be quite large in old-growth forests, and, in the boreal, the bryophyte pool as well. The conversion of old-growth to managed forests likely results in a loss of ecosystem carbon that can never be regained. Protection of old-growth thus constitutes a legitimate carbon sequestration strategy.

### Policy considerations

- The concept of carbon additionality is central to carbon credit and offset schemes. It is difficult to demonstrate additionality in most forest management practices. By its nature, forestry often causes reductions in carbon stocks, especially from the vegetative pool. But a contribution can still be made to climate change mitigation by adjusting these practices so as to *minimize carbon release* as opposed to *maximizing carbon sequestration*. The former idea is gaining traction through such mechanisms as offsets for reduced deforestation/degradation and reduced impact logging. If boreal and temperate forests are to be included in a carbon credit and offsets scheme, it will likely be necessary to recognize such contributions, which are potentially more feasible than “traditional” carbon additionality.
- If policy makers choose to include such “reduced carbon release” practices in a credit/offset scheme, they will need to set a baseline that allows these practices to demonstrate additionality. If the baseline is a natural, unmanaged forest, then most forest practices will always appear carbon-negative. But if the baseline is a “business-as-usual” managed forest, then such practices will constitute a creditable improvement over the baseline. Setting baselines is not a purely scientific process; it is an act of policy that determines which forest management activities will be incentivized.
- The practice of extending rotations offers a straightforward biological means of increasing carbon sequestration in existing forests, and thus has become a

focus for forest managers participating in carbon offset markets. It has been suggested that carbon offset credits can be used to produce a large-scale dividend of additional carbon sequestration by subsidizing landowners to extend rotations until peak stand productivity (in silvicultural terms, when periodic annual increment and mean annual increment are equal) (Wayburn, 2009). In this way, carbon “density” on an individual acre will be increased by allowing forests to more closely approach their natural productive potential.

However, the well-known market externality of “leakage” complicates the implementation of this concept. If revenues from carbon credits motivate enough landowners to extend rotations, then a “hole” of sorts opens up in the wood supply. The landowners may well plan to harvest the same (or greater) volume several decades from now, but that does nothing to change the current demand for wood. Mills will be forced to increase the price they pay for roundwood, which will likely motivate landowners not participating in carbon sequestration activities to cut and sell more wood than they otherwise would have (and perhaps *earlier* in the rotation than they planned). Thus, while some landowners delay harvesting in order to accumulate more carbon per forested acre, other landowners will accelerate harvest to fill the gap. While the former landowner will in the end produce *more* wood per acre than he or she would have without the extended rotation, we must also consider the growth *foregone* from the latter land-owner’s forest, which was cut earlier than it otherwise would have been.

This example is intended to illustrate the difficulty of preventing leakage when demand for wood remains constant (or grows).

- Another important policy factor is whether to consider long-term forest products as a carbon pool. The choice could well determine whether or not practices like thinning are positive, neutral or negative from a carbon sequestration perspective. If the carbon contained in forest products is “sequestered,” then a great many more forestry projects would be eligible for carbon credits and offsets than if that carbon is “released.” The designers of offset systems will need to balance the increased measurement and documentation burden of including a forest products carbon pool with the potential to include more projects.
- This review indicates that many forest practices have a minimal impact on the soil carbon pool, which is the most difficult to measure. Thus, it may be possible that offsets involving certain forestry practices could go forward without strict quantification of this pool. This would considerably reduce measurement cost. As a rule, quantification would likely be least vital when the practice in question results in minimal soil disturbance.

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## Chapter 11

# Managing Afforestation and Reforestation Projects for Carbon Sequestration: Key Considerations for Land Managers and Policymakers

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### EXECUTIVE SUMMARY

Forest management of planted and natural secondary forests for carbon sequestration, applied in the appropriate contexts, presents many opportunities for climate change mitigation and adaptation.

In climate change policy discussions, planted and natural secondary forests are placed in the category of afforestation and reforestation (A/R) projects. Temperate regions currently contain most of the existing planted and naturally regenerating forests. However, establishment of new forests is fastest in the tropics, especially Southeast Asia and Latin America.

We identify two key success factors for A/R projects in general, and for carbon sequestration in particular:

- **Site selection.** In order to manage A/R projects for carbon sequestration successfully, managers must select appropriate sites. Selecting the right site can result in forests that are both productive and efficient at sequestering carbon. In particular, it is important to understand how new forests will affect soil carbon reserves. On inappropriate sites, A/R projects can result in losses of soil carbon that are in conflict with the objective of sequestering carbon. In addition, newly established forests can affect water quantity, water quality, and biodiversity. While opportunity exists for carbon sequestration projects, carbon should not supplant all other forest values.

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Rather, managers should treat carbon as one of many management objectives for forests.

- **Species selection.** Selecting species that are appropriate for site conditions and management objectives is necessary for a successful A/R project. Mixed-species forests, containing species that occupy different ecological niches on the same site, have the potential to store more biomass, and therefore carbon. Single-species forests are less complex to manage, and often benefit from years of research and phenotypic selection, resulting in high growth rates and carbon sequestration. Therefore, while mixed-species forests have great potential, the extensive research and knowledge regarding single species forests often leads to more certain timber production and carbon sequestration.

After managers select an appropriate site and species mix for the desired outcomes of the project, active forest management can increase carbon sequestration incrementally. We review common forest management practices and their effect on forest carbon sequestration, including:

- **Site preparation.** Generally, site preparation increases root and tree growth, improving biomass production. However, site preparation can cause loss of soil carbon and inherently involves significant fossil fuel emissions.
- **Fertilization.** When managers supply the proper nutrients to a forest in the proper amounts, fertilization increases carbon sequestration. Fertilization also results in greenhouse gas emissions due to the fertilizer production and application process. Alternatives to fertilizer include planting nitrogen-fixing species in A/R projects.
- **Irrigation.** Irrigation can dramatically increase forest growth rates, but may be prohibitively expensive or impractical.
- **Herbicides.** Controlling competing vegetation with herbicides produces the best results when applied as part of site preparation. After A/R projects fully occupy a site, there is little benefit to carbon sequestration from herbicides.
- **Thinning.** Selectively harvesting individual trees, commonly called thinning, always has a negative short-term impact on forest carbon stocks. However, thinning improves timber quality and tree vigor and can reduce the risk of a reversal of carbon sequestration due to fire, windthrow, insect infestations and disease.
- **Harvesting.** Forest managers can increase carbon stocks by reducing logging impacts on residual trees and the forest floor. Increasing rotation lengths and retaining logging slash on site can also increase carbon stocks.

Based on our review and analysis of the peer-reviewed literature on afforestation and reforestation, here are the key implications we see for forest managers and policy makers:

- Afforestation of sites that have historically not supported forests generally has adverse effects on ecosystem values other than carbon sequestration. Policy makers should consider whether their incentives for forest carbon should promote this type of activity.
- Forest managers should consider using nitrogen-fixing species in place of fertilizers. This can result in reduced emissions from fertilizer production and increased forest biomass.
- Thinning, while reducing short-term carbon sequestration, is an important management technique to reduce the risk of forest loss, improve long-term carbon sequestration, and improve timber quality.
- A/R activities often involve site preparation and/or soil disturbances, which affect soil carbon sequestration. Soil carbon often represents a significant portion of total ecosystem carbon; therefore, policy makers should include soil carbon pools in A/R carbon legislation to avoid unintended carbon emissions.
- Policy makers should consider how to incentivize or protect other ecosystem services besides carbon to help ensure that unintended negative side effects of A/R projects do not ensue.
- Large, industrial, single-species plantations developed by institutional investors dominate A/R projects. Policy makers should seek ways to make native and mixed species plantations economically competitive with single species systems because, in some cases, they offer additional carbon storage and reduced risk of carbon loss from pests and disease.
- Additional research is needed on the management of native species in tropical countries.
- Long-term carbon sequestration studies of A/R projects are lacking. It is important to monitor existing projects as they progress into older forests.

**Keywords:** Afforestation, reforestation, carbon sequestration, silviculture

## INTRODUCTION

Deforestation and forest management account for an estimated 17% of global greenhouse gas (GHG) emissions (IPCC, 2007). As a result, forests and forest management are receiving significant attention in both domestic and international climate change policy discussions (Angelsen, 2008; Broekhoff, 2008). The Clean Development Mechanism (CDM) of the Kyoto Protocol currently includes afforestation and reforestation<sup>1</sup> (A/R) projects; however, only three such projects have been approved by the CDM board as of May 2009 (UNFCCC, 2009). In addition to A/R projects, policy makers are now considering including carbon credits from

<sup>1</sup> For the purposes of this analysis, we define reforestation as planting or natural regeneration of forest on land that previously supported forest (i.e. planting trees on cropland, which supported forest prior to land clearance). We define afforestation as planting trees on land that has never previously supported forest (i.e. planting trees on steppe or pampas grassland ecosystems that do not naturally support forest).



Reduced Emissions from avoided Deforestation and forest Degradation (REDD) under a successor to the Kyoto Protocol. Various voluntary carbon registries and sub-national level programs also include forestry to some degree (for more on the topic of global policy, see Chapter 18, this volume).

While forestry has received much attention as a low cost source of emission reductions, a key success factor for forest carbon projects is high-quality management. Land managers will need to understand both the science of how forests grow and sequester carbon, and the communities and people associated with forests. In this chapter, we review the silviculture (the science of managing forests) of afforestation and reforestation projects as it relates to carbon sequestration.

First, we present some of the trends in planted and secondary forests across the globe. We then discuss key concepts for reforestation and afforestation projects. Next, we review the suitability of different sites for A/R projects. Assuming a site is suitable for A/R, we then discuss species selection. Finally, we review how some of the most common silvicultural treatments affect the carbon balance of A/R projects. Throughout, we illustrate the management of A/R projects for carbon sequestration with two case studies: reforestation with *Eucalyptus* spp. in Brazil, and *Acacia* spp. in Indonesia.

It is our hope that this chapter will be instructive for foresters managing A/R projects for carbon sequestration under different circumstances, and help policy makers develop appropriate and effective forest carbon offset legislation.

### **Global afforestation/reforestation trends**

To understand the trends in afforestation and reforestation on a global scale, first it is helpful to define different types of forests. Primary forests are those forests that have never been cleared and have developed under natural ecological processes. Secondary forests are those forests that have regenerated by natural processes following the clearance of primary forests or a change in land use, for example, to agriculture, and then abandonment and reversion back to forest. Plantations are forests that humans have planted either on landscapes that once supported primary forest or on land that did not previously support forest. Plantations may be established using native or exotic species, or a combination of both. Afforestation projects are always plantations, while reforestation projects may be plantations or secondary forests.

### **Historic patterns of forest cover based upon site productivity**

Variation in soil quality and productive capacity drives the distribution of land use across the globe. The inherent productive capacity of land has led to common processes of land colonization and abandonment in areas experiencing afforestation and reforestation. This phenomenon has been identified by Mather and Needle, (1998) as the “Forest Transition,” which they characterize as an adjustment of agriculture to site quality and inherent productivity.

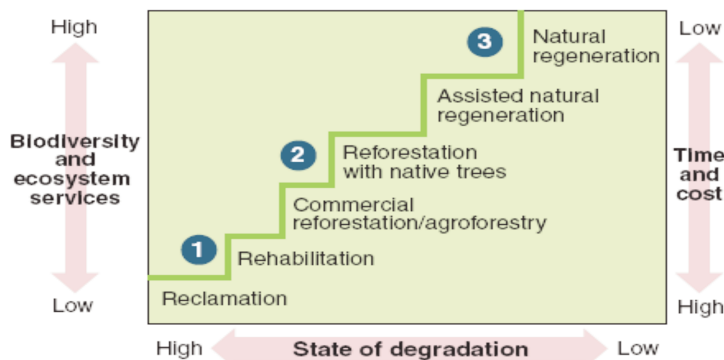
In the early stages of a human colonization, vast areas of forest are cleared for agriculture. As a society industrializes and urbanizes, marginal lands are abandoned and agriculture is concentrated on the most productive sites. Agricultural abandonment typically follows one of two pathways: scarcity of employment in rural areas leading to migration to urban areas, or scarcity of forest products to meet demand. These pathways result in different types of secondary forests. Scarcity of employment (currently in Europe and the Mediterranean) results in more naturally regenerating forests, while scarcity of forest products (currently in SE Asia and Latin America) results in more intensively managed plantations (Rudel et al., 2005). This phenomenon has been observed throughout the temperate regions, and similar processes are beginning in the tropics (Rudel et al., 2002).

Depending on access to markets and the economics of competing land uses, deforestation and reforestation may occur simultaneously in the same region (Sloan, 2008). In Panama, for example, reforestation has begun in many parts of the country, while deforestation continues in others. International forestry companies and investors are responsible for most of the reforestation occurring in Panama (Sloan, 2008). As well-capitalized forestry firms convert pasture to plantations, ranchers and farmers move to new frontiers and continue deforestation.

As marginal agricultural lands are abandoned, they often transfer to pasture and then to forest, or directly to forest. This creates an opportunity for A/R as countries industrialize. The specifics of a given A/R project depend on site quality and access to markets – more intensive silviculture is practiced on the more productive abandoned land, while natural regeneration is often more practical on low productivity and remote sites.

If land has been degraded, natural regeneration is often impossible or impractically slow (Figure 1) (Chazdon, 2008). Infrastructure, roads, access to international timber markets and human capital in the form of professional foresters, all make intensive forest management more economical. If these elements are absent, it is more likely that A/R will take the form of natural regeneration.

**Figure 1 Restoration staircase of previously forested landscapes**



Source: Chazdon, R.L., 2008. Beyond deforestation: Restoring forests and ecosystem services on degraded lands. *Science* 320, 1458-1460. Reprinted with permission.

### Current forest cover patterns

Global primary forest area has declined from 1,397 million hectares in 1990 to 1,337 million hectares in 2005, or a loss of approximately 4 million hectares of primary forest per year (FAO, 2006a). The rate of primary forest loss is accelerating, and accounts for the majority of global forest losses from 2000-2005 (Table 1).

**Table 1 Total global forest areas, 1990-2005 (thousands of hectares).**

	1990	2000	2005	Annual Rate of Change	
				1990-2000	2000-2005
<b>Total Forest Area</b>	4,077,291	3,988,610	3,952,025	(8868)	(7317)
<b>Total Primary Forest Area</b>	1,397,585	1,373,536	1,337,764	(2405)	(7154)
<b>Total Plantation Area*</b>	102,636	126,938	139,772	2,430	2,567
<b>Other Forest - Including Secondary</b>	2,577,070	2,488,136	2,474,489	(8893)	(2729)

\* Plantation Area in this table only includes planted exotic species. Table 2 includes native and exotic planted forests.  
Source: FAO Global Forest Resources Assessment, 2005. Authors' analysis.

The loss of primary forests is especially disturbing from a global carbon balance perspective. Primary forests have been shown to contain more carbon than the secondary forests, plantations, agriculture, agroforestry systems and pastures that replace them. Therefore, if reducing greenhouse gas emissions through forest management is an objective of society, the first priority should be to minimize the loss of intact primary forest.

In contrast, plantations, of both native and exotic species, compose an increasingly large proportion of global forest area. Global plantation area increased from 209 million hectares in 1990 to 271 million hectares in 2005, equating to 4.1 million hectares of new plantations per year (Table 2). While these rates of primary forest loss and new plantation establishment are similar in magnitude, it should not be inferred that primary forest is being converted directly to plantation forests, although this may be true in some regions.

**Table 2 Total planted forest area. Includes exotic and native species (thousands of ha).**

Region	Total Planted Forests*		
	Area		
	1990	2000	2005
Africa	13,783	14,371	14,838
Asia	100,896	114,820	131,981
Europe	68,400	76,328	79,394
North and Central America	14,758	26,084	29,050
Oceania	2,447	3,491	3,865
South America	9,157	11,462	12,215
<b>Total World</b>	<b>209,441</b>	<b>246,558</b>	<b>271,341</b>

\*Includes planted native species and planted exotic species

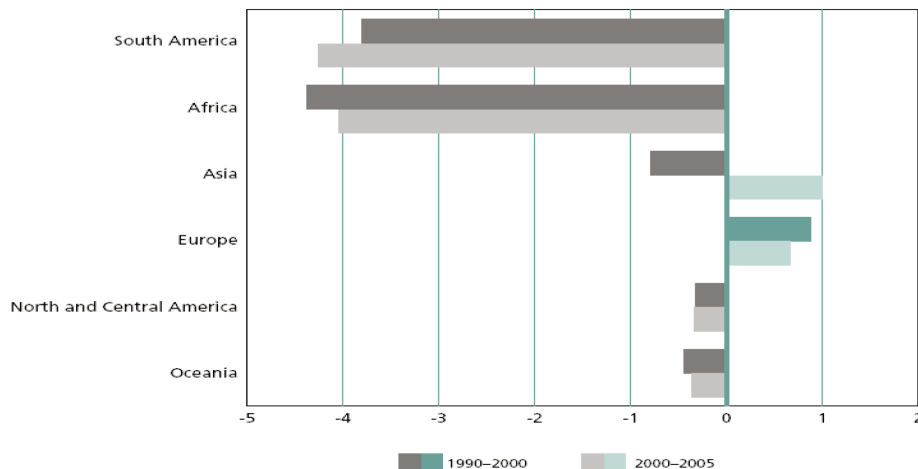
Source: FAO. 2006. Global planted forests thematic study: results and analysis

The loss of primary forests is especially disturbing from a global carbon balance perspective. Primary forests have been shown to contain more carbon than the secondary forests, plantations, agriculture, agroforestry systems and pastures that replace them (Montagnini and Nair, 2004; Gibbs et al., 2007). Therefore, if reducing greenhouse gas emissions through forest management is an objective of society, the first priority should be to minimize the loss of intact primary forest.

Although the exact area is unknown, naturally regenerating secondary forests compose a significant portion of the forests under the A/R umbrella, (FAO, 2006b). Given the young age of many of the planted and naturally regenerating forests that have established globally since 1990, they are likely sequestering large amounts of CO<sub>2</sub> from the atmosphere.

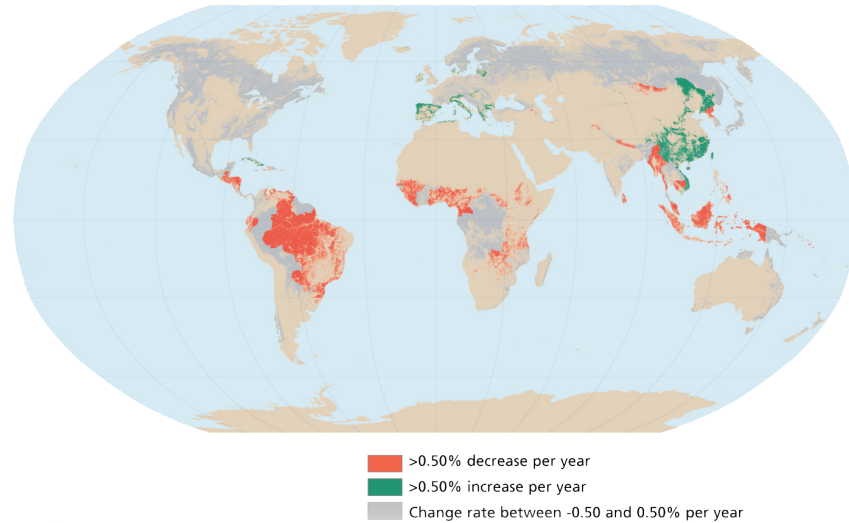
Planted forests are following different trends in different regions of the world. Asia has the largest area of planted forests, followed by Europe and the Americas (Table 2, Figures 2, 3). The FAO classifies planted forests by their primary purpose (production or protection) and species. *Pinus* (pine species) is by far the most commonly planted genus. *Acacia*, *Eucalyptus* and *Cunninghamia* (Asian fir) also represent large components of global planted forests (Table 3). *Acacia*, *Eucalyptus* and *Tectona* (teak) are tropical species, while the other commonly planted genera are temperate species. This suggests that while A/R is becoming more prevalent in tropical countries, there is still much more land area of A/R in temperate regions. We do not present the distribution of species by region here, but it is available in FAO's Global Planted Forests Thematic Study (FAO, 2006b).

**Figure 2 Annual net change in forest area by region 1990-2005 (millions of hectares per year)**



Source: FAO, 2006. Global forest resources assessment 2005: Progress towards sustainable forest management. FAO Forestry Paper 147.

The extent and high growth rate of planted forests has generated interest in using A/R projects as a means of carbon sequestration. While the rate of establishment of A/R forests has increased in recent years, there are still large areas suitable for A/R projects. The IPCC estimates that the potential exists for 345 million hectares of new plantations and agroforests (Cannell, 1999). In addition, many policy makers and foresters point to the positive effects A/R can have on ecosystem services such as water, soil quality and biodiversity (Plantinga and Wu, 2003; Cusack and Montagnini, 2004; Schoeneberger, 2005; Carnus et al., 2006).

**Figure 3** Net change in forest area 2000-2005

Source: From FAO, 2006. Global forest resources assessment 2005: Progress towards sustainable forest management. FAO Forestry Paper 147. Reprinted with permission.

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Soil carbon stocks directly affect the carbon balance of A/R projects; therefore, they should be included in a carbon offset program. The impacts on other ecosystem services, while not directly related to carbon sequestration, are tradeoffs that land managers and policy makers will need to evaluate.

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On the other hand, many ecologists, soil scientists and foresters have raised concerns over certain A/R projects that may cause loss of soil carbon (Farley et al., 2004; Hirano et al., 2007) or reduced stream flow and water yield (Scott and Lesch, 1997; Farley et al., 2005; Wang et al., 2008). Soil carbon stocks directly affect the carbon balance of A/R projects; therefore, they should be included in a carbon offset program. The impacts on other ecosystem services, while not directly related to carbon sequestration, are tradeoffs that land managers and policy makers will need to evaluate.

Before reviewing the literature on appropriate locations for A/R projects, we first introduce some key concepts that are essential for successful A/R implementation and management.

**Table 3** Global plantation area by species in 2006.

	Plantation Area (1000 ha)	
	Productive	Protective
<i>Acacia</i>	7,357	1,554
<i>Eucalyptus</i>	11,981	1,693
<i>Cunninghamia</i>	15,393	770
<i>Picea</i>	6,284	867
<i>Pinus</i>	46,067	8,802
<i>Populus</i>	4,241	4,949
<i>Tectona</i>	5,819	20
All Others	44,794	29,775
Total	141,936	48,430

Source: FAO. 2006. Global planted forests thematic study: results and analysis

## KEY CONCEPTS

### Forest ecosystem carbon stocks and flows

Globally, terrestrial ecosystems, including forests, are estimated to sequester between 1.8 and 3 billion tons of carbon dioxide (CO<sub>2</sub>) annually (Dixon et al., 1994; Canadell and Raupach, 2008). To manage forests of any type for carbon sequestration, it is important to understand how carbon is stored and cycled through a forest ecosystem. Very broadly speaking, carbon is present in four pools: above ground biomass, below ground biomass, dead woody debris, and soil carbon. Above ground biomass includes all tree and plant parts including the tree stem, branches, and leaves. Below ground biomass includes both coarse and fine plant roots. Dead woody debris includes decaying biomass on the forest floor such as leaves, branches and entire trees. Soil carbon includes the organic matter incorporated into the soil itself. Carbon flows between these sinks and the atmosphere in a complex manner, described in more detail by Dixon et al. (1994) and (Malhi et al., 1999). A more in-depth review of forest stand dynamics in relation to carbon sequestration is provided in Chapter 3 of this volume.

### Additionality

A common principle underlying carbon offset projects and protocols is additionality over a specified baseline. In other words, to be awarded offsets, a project must demonstrate that the carbon it sequesters is beyond what would have happened in the absence of the project. This principle applies to A/R projects as well, and is particularly important when considering site selection for A/R projects.

### Degraded forests

The term “degraded” is used loosely in describing forests impacted by human activity or management, and is used to justify converting land to plantations. Here we distinguish between structural and functional degradation, and suggest that structurally degraded forests are still functioning forests and therefore should not be eligible for A/R funding. In contrast, functionally degraded sites are no longer able to successfully support trees and are therefore legitimate sites for restoration and A/R.

Structural degradation usually entails small scale but continuous chronic site disturbance that alters the species composition or structure of the forest canopy. Functional degradation is usually the result of acute, one-time disturbances, which alter the site productivity and physical characteristics of the soil (Ashton et al., 2001).

Heavily logged primary forests, with little remaining valuable timber, are often regarded as structurally degraded. While merchantable timber may be lacking, many cutover forests continue to grow and serve as carbon sinks and wildlife habitat. Furthermore, recently cutover forests are not non-forested land and therefore should not be eligible for A/R funding because it will create perverse incentives to high-grade natural forests, classify them as degraded, and then replace them with plantation forests. Rather, structurally degraded forests should be considered under REDD or Improved Forest Management methodologies. In contrast, functional degradation

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Here we distinguish between structural and functional degradation, and suggest that structurally degraded forests are still functioning forests and therefore should not be eligible for A/R funding. In contrast, functionally degraded sites are no longer able to successfully support trees and are therefore legitimate sites for restoration and A/R.

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alters forest sites to such a degree that trees can no longer grow on the site and active reforestation is often the most efficient and practical means to restore forest to the site (Parrotta, 1990, 1991; Lamb et al., 2005). Examples of functionally degraded sites are those that have been used for surface mining, or intensive agriculture and pasture. Such sites when abandoned have lost the capacity to naturally re-vegetate to forest because: i) no viable seed source for natural regeneration exists; ii) the hydrology and the fertility of surface soil horizons have been altered to an extent that cannot allow seed germination or establishment of trees; or iii) degradation has allowed opportunistic vegetation to colonize that is maintained by new cyclical disturbances (e.g. fire).

### **Risk aversion**

In addition to carbon sequestration, forests should be managed to minimize the risk of carbon loss through disturbance. Depending upon site productivity, managers can assume different levels of risk in their management strategy. High productivity sites support shorter rotations and encourage managers to practice more intensive, expensive management. On high productivity sites, there is less chance of disturbance over the short rotations and the pay-off is greater at the end of the rotation. Even if there is a disturbance, the rotations are short enough that one can easily replant and start over.

Forests on marginal lands grow more slowly and therefore landowners must wait longer to derive a useful product. Longer rotations expose forest stands to disturbance (disease, fire, wind throw) for greater periods, making the loss of some or the entire timber crop more likely. In addition, long rotations result in lower rates of return, all else being equal, because cash flows are realized further in the future.

Therefore, managers generally practice less capital-intensive silviculture as they move to progressively less productive sites. This is supported by the land use trends that can be observed on the landscape and in the theoretical models of forest management (Mather and Needle, 1998). Natural regeneration and passive management for carbon sequestration may be more appropriate on marginal lands, although land managers will need to conduct their own economic and silvicultural analyses for their specific site.

### **Management intensity – *Acacia mangium***

*Intensive management of A. mangium plantations on rich sites in Indonesia (fluvisols) has been found to maintain high production levels of carbon and/or timber over successive rotations without significant loss of nutrients (Mackensen and Folster, 2000). A. mangium plantations on poorer sites (arenosols, acrisols, ferralsols), however, cause nutrient losses that threaten the long-term productivity of the site and that can only be compensated for with expensive investments in fertilizer. Thus, site productivity needs to be considered in deciding how intensively to manage a site.*

### Promoting resiliency against disturbances

Practicing sound silviculture to promote resiliency and minimize risk of major disturbances is key when managing for carbon sequestration. A major stand-clearing disturbance such as a fire can release most or all of a forest's aboveground carbon stocks, reversing any carbon sequestration benefit. Protecting a stand against disturbance can involve management practices that reduce a stand's aboveground carbon stocking, such as thinning. Management that slightly reduces carbon stocks in the short-term is worthwhile when it helps avoid the types of disturbances that can wipe out a forest carbon offset project.

Thinning stands in fire-prone regions is one of the most effective means for reducing the risk of catastrophic fire (Finkral and Evans, 2008). We discuss the specifics of how thinning affects forest carbon sequestration in greater detail below. Maintaining a mix of species can also be effective in lessening the potential damage from disturbances that target a particular species, such as insect outbreaks (Jandl et al., 2007).

### SITE SELECTION FOR AFFORESTATION/REFORESTATION PROJECTS

The most important management decision for a successful afforestation/reforestation project is selecting an appropriate site. The addition of trees to a non-forested site will increase above-ground carbon storage in almost all cases. A/R projects vary in how they impact soil carbon, water, and biodiversity, and adverse impacts to any of these forest values must be considered when deciding where to site an A/R project.

#### **Risk: *Acacia mangium***

*Acacia mangium* accounts for approximately 80% of short-rotation plantations in Indonesia. Incidence of heartrot fungi in these stands is as high as 46.7% in some regions of the country (Barry et al., 2004). Root rot is also prevalent (as high as 28.5% incidence) in southeast Asian *A. mangium* plantations, particularly in second and third rotations. Root rot, however, was found less often in former grasslands than in lowland former rainforest. Further, waiting for two months between harvesting and replanting was found to reduce the incidence of root rot (Irianto et al., 2006). Using a mixed species approach can help diversify the investment in reforestation, so that if a plantation becomes heavily infected, not all trees are lost. Forest fire poses an additional risk in Indonesian *Acacia mangium* stands. The high litter fall produced by *A. mangium* combined with dry conditions and *Imperata* grassland understories has caused significant losses of forest to fire (Saharjo and Watanabe, 2000).



### Impact on soil carbon

While most A/R projects will increase aboveground carbon stocks (because trees tend to store more carbon than other types of land cover, namely, shrubs, grasses, or crops), they will not necessarily increase soil carbon stocks. In grasslands, carbon accumulates in the soil each year as grasses die and decompose. If a forest replaces grassland, the tilling and site preparation necessary to plant new trees exposes the soil carbon to increased levels of oxygen. This speeds up soil carbon decomposition rates, and carbon dioxide emissions.

A similar phenomenon occurs if peat is drained to improve site conditions to plant forests. Peat is generally very moist, and creates an oxygen-poor environment where decomposition happens very slowly. Draining peat increases the oxygen levels in peat soils, resulting in faster decomposition and carbon dioxide emissions (Jaenicke et al., 2008).

Given that approximately 75% of terrestrial carbon is stored in soils (Paul et al., 2002), it is vitally important to monitor soil carbon as well as aboveground carbon for A/R projects. Afforestation projects – projects on land that has never previously supported forests – run the biggest risk of causing large soil carbon releases. A/R projects may also cause changes to other environmental services such as water runoff and biodiversity. Managers should weigh these potentially negative changes against the benefits of carbon sequestration in deciding whether to initiate an A/R project.

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### *Agricultural land*

Abandoned agricultural land is perhaps the most common land cover type for A/R projects. Agricultural land is found across a wide range of ecological settings and can encompass land used for crops as well as for pasture, making it difficult to generalize about its suitability for A/R Projects. In this section, we attempt to differentiate between some of the different types of agricultural land, and to assess their suitability for A/R projects.

In general, afforestation of cropland has been found to increase soil carbon content in the long-term, following an initial decrease. In contrast, afforestation of pastures has been shown to slightly decrease soil carbon (Paul et al., 2002). However, these overall trends vary by region and forest type. A/R on tropical or subtropical sites results in greater soil carbon stocks than using similar species in temperate regions, and deciduous species result in greater soil carbon gains than evergreen species. With both crop and pastureland, management intensity affects how soil carbon stocks change. Letting a secondary forest grow on former crop or pasture land often results in greater soil carbon levels than a plantation because there is less soil disturbance (Guo and Gifford, 2002).

Afforestation or reforestation affects pastureland (potentially arable grassland) soil carbon stocks in various ways depending on specific site conditions. One key source of variability is precipitation. The A/R potential of drier grassland sites needs to be considered separately from the A/R potential of wetter sites. Afforestation on arid or semi-arid grasslands has been shown to create carbon additionality, although care

must be taken to select species that are efficient in their water use. In Inner Mongolia, poplar (*Populus spp.*) and Mongolian pine (*Pinus sylvestris var. mongolica*) have been used to afforest semi-arid grasslands. Soil carbon under poplar plantations recovers to pre-afforestation stocks by age 15, while soil carbon under Mongolian pine persists below the pre-afforestation grassland levels after 30 years. Although soil carbon decreased under Mongolian pine, significant increases in aboveground and belowground (root) carbon stocks resulted in both pine and poplar stands being net carbon positive (Hu et al., 2008).

Afforestation of pasture land in the Patagonian semi-arid steppe has also resulted in net carbon sequestration (Laclau, 2003a; Noretto et al., 2006). The grass-shrub steppe of Patagonia stores approximately 95.5 Mg C/ha, predominantly as soil carbon (Laclau, 2003a). Afforestation with exotic Ponderosa pine (*Pinus ponderosa*) resulted in no loss of soil carbon and significant gains in aboveground biomass after 14 years. Naturally regenerated native cypress (*Austrocedrus chilensis*) stands, with an average age of 45 years, showed significant increases in soil carbon (Laclau, 2003a) and even greater total carbon storage than Ponderosa pine. The different average stand ages in this study make comparison of the rates of carbon storage between the exotic ponderosa pine and native cypress difficult. However, given that cypress regenerates naturally in the steppe ecosystem, it is likely a more efficient method of long-term carbon storage than planted Ponderosa pine.

Afforestation of grasslands in wetter climates has greater potential to release large amounts of soil carbon. For instance, in the Ecuadorian highlands, radiata pine (*Pinus radiata*) has been used to afforest grasslands on carbon-rich, volcanic soils in a relatively wet climate (Farley et al., 2004). The wet, oxygen poor soils store large amounts of carbon. When these wet grasslands are drained and exposed to oxygen, rapid decomposition of soil carbon occurs. In twenty-five year old plantations of radiata pine in the Ecuadorian highlands, carbon stocks were reduced in the first 10 cm of soil from 5 kg/m<sup>2</sup> under native grasslands to 3.5 kg/m<sup>2</sup>. Soil carbon content decreased at greater depths as well.

In contrast, reforestation of grasslands that were once tropical forest has the potential to increase soil carbon storage. In Indonesia, Imperata grasslands now cover 8.5 million ha of what was once primary forest (van der Kamp et al., 2009). Secondary forest growth has the potential to store 61.7 tons/ha (East Kalimantan) to 219 tons/ha (Sumatra) of carbon as compared to Imperata grassland baselines of 39.64 tons/ha (East Kalimantan) and 47 tons/ha (Sumatra) (van der Kamp et al., 2009).

### *Peatland*

While A/R on peatland is not as common as on agricultural land, the large amounts of carbon stored in peatland soil warrants discussion. The carbon-rich peatlands of Southeast Asia have increasingly become a target for drainage and conversion to plantations. Although peatlands only comprise 3% of the world's land area, they store one-third of the world's soil carbon (Rydin and Jeglum, 2006). Impacts of afforestation on peatland soil carbon differ depending on where the project is located.

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The impacts of afforestation on peatlands depends upon the depth of peat (and hence the amount of drainage required) as well as the climate. In colder climates such as the UK or Scandinavia, shallow peats requiring less drainage result in lower levels of soil carbon release, and afforestation projects may be net positive (Hargreaves et al., 2003; Byrne and Farrell, 2005; Byrne and Milne, 2006). For instance, afforestation of peatland in Britain in peat less than 35.5 cm deep resulted in an increase in aboveground biomass that could compensate for the loss of soil carbon of 50-100g C/m<sup>2</sup>/yr (Cannell et al., 1993). In deeper peat in Britain, where carbon release from drainage can reach 200-300g C/m<sup>2</sup>/yr, aboveground biomass did not compensate for the loss in soil carbon (Cannell et al., 1993). Peatlands in Southeast Asia are deeper and store considerably more carbon than those in the UK or Scandinavia. Indonesian peatlands have been estimated to store 55 Gt C (Jaenicke et al., 2008). One recent study of carbon release from a drained peat swamp estimated an average of 313-602 g C/m<sup>2</sup>/yr released over three years (Hirano et al., 2007). Given these high levels of soil carbon release, policy makers and managers need to look closely at afforestation projects on tropical peatlands to ensure additionality.

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### **Carbon in relation to other forest/land management values**

If land managers and policy makers determine that an A/R project provides carbon additionality, they should then evaluate the carbon benefits alongside other important forest and ecological values. Ideally, A/R projects will provide carbon benefits as well as economic and ecological benefits – increases in biodiversity, water quality, etc. – and the decision to proceed will be straightforward. Managers will face difficult choices when A/R runs the risk of having negative impacts on other ecological values, even while providing carbon additionality.

### *Water*

Afforestation on grassland and shrubland can alter the hydrology of a system by decreasing runoff and increasing transpiration. This can be particularly problematic in drier locations where water limitation is an issue. Globally, grassland and shrubland afforestation have been found to reduce annual runoff by as much as 44% and 31%, respectively, for up to 20 years after afforestation (Farley et al., 2005). Fast-growing species demand more water and will induce greater water flow reductions (Bruijnzeel et al., 2005). Studies conducted in South America have demonstrated significant decreases in water levels, both in the drier steppe (in Patagonia, using Ponderosa Pine and native Cypress), as well as in the wetter pampas (in Argentina, using *Eucalyptus camaldulensis*) (Engel et al., 2005; Licata et al., 2008). In addition, it was found that afforestation of the Argentine pampas with *E. camaldulensis* acidifies the soil, reduces the soil cation exchange capacity (Jobbagy and Jackson, 2003), and results in soil and groundwater salinization (Jobbagy and Jackson, 2004). Therefore, while afforestation may enhance short-term carbon sequestration, it also alters soil chemistry in ways that can significantly reduce future productivity and impact groundwater quality.

Reforestation, like afforestation, can also reduce overall water flow due to the demand for water by trees (Bruijnzeel et al., 2005). Whereas afforestation reduces water flow to levels that the site may not be adapted for, reforestation reduces water flow to levels similar to when the site was previously forested (Bruijnzeel et al., 2005). Further, reforestation can have other positive side effects in relation to water values. Deforestation has been shown to increase the risk and severity of flooding (Bradshaw et al., 2007), and reforestation has been proposed as a means of mitigating flood risk. In the Panama Canal Watershed, for example, reforestation moderates the variation in water yield, especially during the dry season (Condit et al., 2001). The Panama Canal uses 2.6 billion m<sup>3</sup> of water annually to fill its locks, with water use relatively evenly distributed across the year. However, the Panama Canal Watershed is located in an area of seasonal rainfall patterns, and the canal's use has been limited during the dry season due to lack of water supply. Deforested watersheds in Panama have high water yields in the wet season, when water is abundant, but low water yields in the dry season when water supply is most limited. In contrast, forested catchments have much higher stream flow in the dry season, when water supply is most limited, while still providing adequate water supply during the wet season (Condit et al., 2001).

### *Biodiversity*

The effect of A/R on biodiversity has been a hotly debated topic. The specific effects of any given project depend heavily on its historical context and location within the broader landscape. Plantation forests almost always provide more suitable habitat for forest species than agricultural land (Brockerhoff et al., 2008). Planted forests can also enhance the matrix between remnant natural forest patches, which has multiple benefits:

- Edge effects on natural forests are decreased;
- Planted forests (depending upon choice of tree species) facilitate dispersal between natural forest patches;
- Forest generalist species often use resources provided by planted forests; and,
- Plantation forests can reduce harvesting pressure on and habitat loss in existing natural forests.

In some cases, A/R increases plant and animal diversity, particularly on degraded lands. As mentioned above, Imperata grasslands have replaced large areas of primary forest in Indonesia. Reforestation of these non-native grasslands with *A. mangium* increased arthropod diversity, although not as much as a naturally regenerating secondary forest (Maeto et al., 2009).

However, at the landscape scale, A/R is not always desirable from a biodiversity perspective. In the case of afforestation, planted forest replaces a natural habitat type (i.e. grassland). If the species that are native to a region depend on grassland ecosystems, afforestation will reduce habitat available for these species and be detrimental to landscape scale biodiversity (e.g. bird diversity following afforestation

in South Africa, (Allan et al., 1997)). In addition, the modeled impacts of afforestation on a South African fynbos site using radiata pine projected a large loss of plant and insect biodiversity (Garcia-Quijano et al., 2007).

### **Biodiversity – Eucalyptus plantations**

*The effect of eucalyptus plantations on biodiversity has been examined in the Brazilian Amazon using forest birds as a biodiversity indicator. A study in the northeast Amazon estimated bird species richness using point count estimates. Primary forest (106.5 species) had greater bird species diversity than secondary forest (70 species), which in turn had greater diversity than eucalyptus plantations (50 species) (Barlow et al., 2007a). Eucalyptus plantations contained almost no species in common with primary forest, and contained very few habitat specialists. Primary forests also contained greater butterfly diversity than secondary forests and eucalyptus plantations, although eucalyptus plantations contained a higher number of individuals (Barlow et al., 2007b). Since reforestation takes place on land that is not forested, the appropriate baseline against which to measure reforestation is pastureland or agricultural land. In São Paulo State, Brazil, Blue-winged Macaws used eucalyptus plantations as habitat, but never used pastureland, coffee plantations or rubber plantations (Evans et al., 2005).*

*Retaining forest strips in the northeastern Amazon, whether riparian or upland, that extend into and through the eucalyptus plantation matrix greatly increases bird diversity (Hawes et al., 2008). Riparian and upland forest strips were found to have species assemblages that closely reflected continuous primary forest. This suggests that while eucalyptus plantations themselves do not contribute significantly to biodiversity conservation, managers can design plantations to contain riparian and upland reserves that do provide significant biodiversity conservation benefits.*

### **SPECIES SELECTION**

A small number of genera comprise much of the global plantation area. Single-species plantations of exotic species such as eucalyptus, pine, acacia, and teak have several advantages that make them popular: they are fast-growing species with known markets; a large body of knowledge exists on their silviculture; and, growing only one species makes for less complicated silviculture than growing multiple species. However, there is a growing body of research indicating that viable alternatives to these single-species plantations exist and can potentially sequester more carbon, while also reducing the risk of carbon loss through disturbance. The following sections address the potential of alternative management approaches to single-species exotic plantations, including mixed-species plantations, native-species plantations, and agroforestry systems.

## Mixed species

Mixed-species plantations can increase carbon storage over single-species plantations by incorporating nitrogen-fixing species and species with complementary light and nutrient requirements. Mixed-species plantations can also reduce the risk of carbon loss from pest and disease outbreaks.

### *Nitrogen-fixing species*

Nitrogen-fixing or leguminous species are typically from the Fabaceae or Leguminosae family, and host rhizobia bacteria on their roots that can convert nitrogen gas,  $N_2$ , into biologically available nitrogen,  $NO_3$  or  $NH_4$ . Growing nitrogen-fixing species such as *Albizia spp.* in combination with conventional plantation species can increase productivity. Nitrogen is often limiting in tropical plantations, and increasing the available nitrogen can increase biomass production and carbon storage (Binkley et al., 1992; Balieiro et al., 2008). For instance, the benefits of nitrogen-fixing species on eucalyptus plantation productivity have been researched extensively in Hawaii. Binkley et al. (1992) found that planting 34% eucalyptus and 66% albizia maximized total biomass on volcanic soils in Hawaii. Not only can total biomass be maximized with the introduction of nitrogen-fixing species, but growth rates of the primary timber species (eucalyptus, e.g.) can be increased as well (DeBell et al., 1997), which can increase revenue from wood products. In addition, high growth rates can be sustained longer into eucalyptus rotations (Binkley et al., 2003). Forrester et al. (2005) attributed benefits to eucalyptus growth from the addition of nitrogen-fixing acacia not only to increases in available nitrogen but also to increased rates of nitrogen and phosphorous cycling.

However, the benefits from nitrogen-fixers vary based upon soil properties (Boyden et al., 2005). If the supply of other nutrients is limited, nitrogen fixers will not necessarily enhance productivity. Also, many leguminous nitrogen-fixers do not grow well in acidic soil conditions (Binkley et al., 1992).

### *Complementary species interactions*

Using mixtures of species can increase plantation productivity if the species are complementary in their use of resources (Kelty, 2005; Carnus et al., 2006). That is, if species have different requirements for light and nutrients, and different growth rates, competition between species may be less intense than within a single species, and total biomass growth on the site can be increased (Forrester et al., 2005). Mixtures of complementary species have been found to maintain productivity at higher densities than single-species plantations (Amoroso and Turnblom, 2006).

Conversely, mixed-species stands can be less productive if the species used are too similar in their requirements. Chen et al. (2003) studied different combinations of mixed-conifer species plantations in British Columbia and found no combinations that were superior to single-species plantations. However, they suggested that strategic selection of shade tolerant and intolerant species mixtures might have produced better results in the mixed-species stands. Performance of mixed species

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There is a growing body of research indicating that viable alternatives to single-species plantations exist and can potentially sequester more carbon, while also reducing the risk of carbon loss through disturbance.

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plantations in central Oregon was also found to vary depending on species composition and the initial spacing of trees (Garber and Maguire, 2004).

### **Risk aversion**

In addition to potential increases in carbon sequestration, mixed-species stands can reduce the risk of significant pest and disease outbreaks, which can release stored carbon. Jactel et al. (2005) concluded, based upon a meta-analysis of single vs. mixed species stands, that damage from insect outbreaks was significantly higher in single-species stands. Mixed-species stands are also less vulnerable to fungal pathogens (Pautasso et al., 2005). In a mixed-species forest, even if one species is attacked by pests or pathogens, another species can replace it and continue to sequester carbon and provide other forest values. In addition, the passive dispersal of disease is slower in mixed species stands (Pautasso et al., 2005).

### **Native species**

Native species provide another potential alternative to exotic single-species plantations. Studies comparing native species to exotic species have shown that some native species have similar or superior growth rates compared to exotic species. Currently, the primary disadvantage of native species plantations is the lack of research and knowledge of their silviculture and wood properties compared with conventional exotic species. However, this is starting to change, particularly in Central America (Carnevale and Montagnini, 2002; Hooper et al., 2002; Wishnie et al., 2007) and Asia (Shono et al., 2007; Thomas et al., 2007). Markets for native species wood products are not as well developed as markets for traditional plantation species, meaning that land owners can be more certain of investment returns for exotic species (Streed et al., 2006).

Even in cases where native species do not generate as much above-ground carbon sequestration as exotics, they can still provide other benefits to long-term carbon storage. In the Nicoya Peninsula of Costa Rica, Piotta et al. (2004) found that pure plantations of teak (*Tectona grandis*, native to Asia) produced more volume per hectare after 68 months than all native species experimental plots, whether pure or mixed, except for pure *Schizolobium parahyba* stands. Teak is also a more valuable wood, with well developed international markets. However, teak is a deciduous species, with its large leaves serving as a fine fuel for ground fires during the dry season. Heavy erosion can occur with the onset of rains when teak is planted on inappropriate sites (Carnus et al., 2006). In one study in Costa Rica, teak also resulted in lower soil organic carbon levels than native species on land converted from pasture (Boley et al. 2009).

### *Native single vs. mixed species*

Much of the native species research has investigated differences between native single-species and native mixed species stands. In the Atlantic coastal lowlands of Costa Rica, mixed native species plantations were found to be competitive with single

native species plantations in terms of biomass production. However, single species stands of *Jacaranda copaia* produced the most biomass (46.6 Mg/ha) after three years (Montagnini and Porras, 1998). *J. copaia* plantations in Costa Rica also promote high levels of seed dispersal and high levels of seed diversity, making this a good species from the perspective of biodiversity and restoration of degraded sites (Zamora and Montagnini, 2007). An interesting finding was that one hectare of mixed plantation (4 species) contained more biomass than the sum of four one-fourth hectare single species plots. This suggests that managers interested in managing multiple species for multiple values or markets should do so in a mixture, rather than separate single species plantations. However, managing mixed stands will require more sophisticated silviculture as well as highly trained foresters.

A study by Redondo-Brenes and Montagnini (2006) of nine native tree species in both single- and mixed-species plantations, found mixed species plantations to be more productive than single species stands, with the exception of one species, *Calophyllum brasiliense*, which stored more carbon in single than in mixed plantations. Pure plantations of *Terminalia amazonia*, *Dipteryx panamensis*, and *Virola koschnyi* stored 83.2, 64.8 and 50.6 Mg C/ha, respectively, whereas a mixed plantation of all three species stored 90.8 Mg C/ha. They also suggest that there may be benefits to planting mixtures of fast and slow growing species due to their complementary use of resources. Fast growing species sequester carbon quickly in the early years, while slow growing species accumulate more carbon in the long term.

In another study in Costa Rica, carbon storage was shown to vary considerably, depending on stand management. Nine to twelve year old single species plantations of *Terminalia amazonia* and *Dipteryx panamensis* contained 55.1 - 79.1 and 36.9 - 91.0 Mg C/ha, respectively, in Sarapiquí, but only 27.5 and 36.5 - 44.4 Mg C/ha, in San Carlos (Redondo-Brenes, 2007). This difference was best explained by stand density, as stands that had silvicultural thinnings stored more carbon than those that had not due to the adverse effects of stand density on tree vigor.

### Agroforestry

While agroforestry systems generally do not sequester as much carbon as primary forests, secondary forests or plantations, they can provide a means of integrating forest carbon sequestration into agricultural production (Montagnini and Nair, 2004). Agroforestry refers to a number of different practices of growing trees on agricultural lands, including alley cropping, riparian buffer strips, silvo-pasture, forest farming, and wind breaks. The inclusion of these systems by smallholders in the tropics could produce significant carbon sequestration. Estimates of carbon storage in agroforestry systems range greatly, from 0.29 to 15.21 Mg C/ha/yr, depending upon site productivity (Nair et al., 2009). Agroforestry systems can also increase soil carbon storage (Haile et al., 2008; Takimoto et al., 2008). Agricultural crops and trees grown together can provide complementary carbon storage benefits, similar to mixed-species plantations. A study by Sharrow and Ismail (2004) found that a silvo-pastoral system in western Oregon sequestered more carbon than a pure plantation or pasture, which they attribute to the complementary nature of the

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Managers interested in managing multiple species for multiple values or markets should do so in a mixture, rather than separate single species plantations. However, managing mixed stands will require more sophisticated silviculture as well as highly trained foresters.

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pasture's soil carbon storage with the trees' biomass storage. However, like site preparation associated with establishment of tree plantations, conversion of land, such as a pasture, into an agroforestry system that requires soil tillage will usually reduce total soil carbon, even with row plantings of trees.

### Species selection summary

Overall, the handful of major plantation species used globally have some significant advantages for carbon sequestration: fast growth rates, a large body of knowledge on how to successfully manage them, existing wood markets, and less complex silvicultural knowledge than is required to grow a native or mixed-species plantation. However, there are situations where alternatives to these single-species plantations can generate increased carbon sequestration. Mixed-species plantations can potentially increase carbon storage over single-species plantations through the integration of nitrogen-fixing trees, and the use of trees with complementary growth patterns (Ashton and Ducey 1997). Mixed-species plantations can also reduce the risk of damage from pests and disease, which is important for ensuring the permanence of carbon storage. Native species also have the potential to be equally if not more productive than exotic species and may be better suited to maintaining the long-term productivity of a site. However, long-term silvicultural research, as well as development of markets, is necessary in order to improve the viability of native species plantations. Finally, agroforestry systems can also store significant carbon on sites where it does not make sense to convert the land entirely from agriculture to forest, but this may not be desirable on pasture or grasslands.

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Growing healthy, well-formed trees through sound forestry practices will produce carbon additionality as well as merchantable timber. This point is particularly important since A/R projects often require a combination of carbon credits and timber sales in order to be economically feasible.

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## MANAGING AFFORESTATION/REFORESTATION FOR CARBON SEQUESTRATION

In this section, we review the carbon balance of A/R forest management in terms of the most common silvicultural treatments that forest managers employ. For each treatment, we present general information regarding how the treatment affects forest carbon balances, with more in-depth case studies on eucalyptus and acacia management. A variety of silvicultural treatments is available to improve tree growth and carbon storage in forests, as well as to minimize risk from catastrophic disturbance. Below we have summarized how silvicultural treatments can influence carbon sequestration in the context of A/R projects.

It should be emphasized that having a knowledgeable manager to oversee an A/R project is more important than any particular silvicultural practice. Growing healthy, well-formed trees through sound forestry practices will produce carbon additionality as well as merchantable timber. This point is particularly important since A/R projects often require a combination of carbon credits and timber sales in order to be economically feasible.

### Pre-planting/site prep

Site preparation includes a variety of operations such as stump removal, mowing, disking, excavating planting pits, ripping, subsoiling, ploughing and control of

competing vegetation. Site preparation has the potential to affect carbon sequestration in three ways. First, site preparation increases the ease with which trees establish and begin growth, accelerating carbon sequestration. Tilling or cultivating the soil prior to planting of eucalyptus increases root growth, uptake of nutrients and water, and initial growth rates (de Moraes Goncalves et al., 2002). The degree to which soils are cultivated prior to planting depends on the specific structure of the soil. Second, site preparation that disturbs the soil exposes soil carbon to oxygen in the atmosphere, which increases CO<sub>2</sub> emissions from soil organic carbon (SOC) decomposition (Jandl et al., 2007). Finally, site preparation is one of the most energy-intensive operations associated with A/R management, and results in significant CO<sub>2</sub> emissions (Table 4). Fossil fuel emissions from site preparation can be reduced if mowing is used instead of disking in clearing/cleaning operations, and if furrowing and ridging are performed instead of ripping and subsoiling (Dias et al., 2007).

**Table 4 Carbon dioxide emissions from typical site preparation, stand tending, and infrastructure establishment operations.**

Operation	CO <sub>2</sub> specific emissions (kg CO <sub>2</sub> ha <sup>-1</sup> )
Stump removal (with digger)	324
Clearing/cleaning	
Mowing	97
Disking	128
Soil scarification	
Excavating planting pits	70
Ripping	235
Subsoiling	117
Ploughing	86
Furrowing and ridging	68
Terrace construction	689
Soil loosening (disking)	39
Selection of coppice stems (with chainsaw)	21
Precommercial thinning (with chainsaw)	27
Infrastructure establishment	
Road building	67
Road maintenance	23
Firebreak building	9
Firebreak maintenance	2

Source: Dias, A.C., Arroja, L., Capela, I., 2007. Carbon dioxide emissions from forest operations in Portuguese eucalypt and maritime pine stands. *Scandinavian Journal of Forest Research* 22, 422-432. Reprinted with permission.

## Fertilization

One of the limiting resources to tree growth and carbon sequestration is nutrient availability. When certain nutrients are unavailable, trees cease to grow at optimal rates. Fertilization is one silvicultural tool available to land managers to increase the biomass production of A/R projects. Applying fertilizer to a stand can increase its

growth rates, and hence, the speed at which it sequesters carbon. Intelligent application of fertilizer requires knowledge of the site and the species in order to know which particular nutrient is limiting growth. As a general rule, phosphorous tends to be limiting in tropical sites while nitrogen tends to be limiting in temperate climates.

### Site preparation – Eucalyptus

*In Brazil, four general soil types have been identified where eucalyptus is planted: sandy, loamy, oxidic and kaolinitic. Soil cultivation can be restricted to the planting holes in well-structured and well-drained soils (sandy or loamy soils), while more intensive site preparation is necessary on compacted or cohesive soils (kaolinitic, oxidic soils) (de Moraes Goncalves et al., 2002).*

Several studies have shown that fertilizer increases aboveground carbon storage in forests (Shan et al., 2001; Sampson et al., 2006; Coyle et al., 2008; Luxmoore et al., 2008). Coyle et al. (2008) measured increases in belowground biomass from fertilization in sweetgum, while Gower et al. (1992) observed reduced litterfall, and reduced mass and production of fine roots. Other studies found that fertilization had no significant impact on soil carbon (Shan et al., 2001; Sartori et al., 2007; Luxmore et al., 2008).

It is widely accepted that fertilizer increases the rate of above and below ground biomass production; however, Markewitz (2006) raised concerns about greenhouse gas emissions associated with fertilizer application. Including fertilizer production, packaging, transport and application in forest carbon budgeting results in 1.48 tons of C emissions per ton of nitrogen fertilizer application (Markewitz, 2006). Therefore, while fertilizer may increase carbon sequestration in forest biomass, there are large emissions costs associated with fertilizer application. These should be considered when evaluating the net carbon balance of plantation systems.

In some cases, inter-planting of leguminous trees and ground covers (e.g. *Desmodium spp.*, *Pueraria spp.*) can be more beneficial to productivity and aboveground carbon sequestration than fertilization (Ashton et al., 1997). For instance, in a study of reforestation on eroded pastureland in Costa Rica, inter-planted leguminous species increased productivity in a native species plantation while fertilizer had no effect (Carpenter et al., 2004).

### Irrigation

Irrigation can also enhance tree growth and aboveground carbon sequestration by providing additional water in moisture limited environments (Gower et al., 1992; Coyle et al., 2008). However, irrigation is a relatively expensive silvicultural treatment; therefore, its cost can only be justified by a high increase in productivity.

### Understory elimination/herbicides

Understory elimination can improve biomass growth of the over-story trees but also removes biomass from the understory. In the Southeast U.S., understory-elimination

and application of herbicide in pine plantations has increased above-ground carbon stores, while at the same time causing net primary production and soil carbon to decrease (Shan et al., 2001, Sarkhot et al., 2007, Sartori et al., 2007).

### Irrigation – Eucalyptus

*Irrigation in E. globulus x urophylla stands in Bahia, Brazil significantly increased plantation growth. Aboveground net primary productivity (ANPP) increased by 18% in irrigated stands in a historically wet year, and by 116% in a normal rainfall year (Stape et al., 2008). The majority of ANPP is concentrated in the bole, suggesting significant gains in timber production with irrigation. In terms of carbon, net ecosystem productivity (ANPP plus below ground NPP, litter and soil carbon fluxes) increased with irrigation from 2.3 to 2.7 kg C/m<sup>2</sup>/yr in the wet year, and from 0.8 to 2.0 kg C/m<sup>2</sup>/yr in the normal year. In terms of efficiency of carbon production, each additional 100 mm of water contributed 0.075 kg C/m<sup>2</sup>/yr in wet years and 0.125 kg C/m<sup>2</sup>/yr in dry years. This suggests that irrigation most efficiently increases net carbon sequestration in dry years.*

### Thinning

Forest thinning, or the selective removal of trees, is a silvicultural technique used to manipulate the spacing between individual trees in a forest stand and improve the growth of the remaining individuals. Thinning causes an immediate loss of carbon from the forest, unless carbon stored in wood products is considered sequestered under carbon offset policies. Multiple studies have measured the reduction in aboveground carbon from thinning, with heavier thinnings resulting in greater reductions (Balboa-Murias et al., 2006; Nilsen and Strand, 2008; Campbell et al., 2009). A/R forests respond to thinning in a common fashion across various sites and species: thinning increases the biomass, thus the carbon content, of individual trees, while reducing the stand level carbon stock (Sayer et al., 2001; Eriksson, 2006; Munoz et al., 2008; Campbell et al., 2009). In a *Eucalyptus nitens* stand in Los Alamos, Chile, stands thinned to 400 stems/ha contained 333 tons/ha of biomass, significantly less than the 437 tons/ha present in stands with 1,100 stems/ha (Munoz et al., 2008). Impacts of thinning on soil carbon are inconclusive, although Selig et al. (2008) measured an increase in soil carbon from thinning in southeastern U.S. loblolly pine plantations.

Carbon sequestration is often only one of many management objectives of A/R projects, and while thinning reduces stand level carbon stocks, it positively affects other stand attributes. Thinning re-allocates growing space to the remaining individuals in the stand, improving their growth rates and quality. This results in higher quality sawtimber and generally increases the economic returns at the end of the rotation. If carbon offsets are awarded for harvested wood products or fuel switching from fossil fuels to renewable biomass energy, the carbon balance of thinning operations may become more favorable (Eriksson, 2006).

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The risk reduction benefits of thinning should be an important consideration in carbon storage projects. Thinning can provide protection against the risk of a major disturbance such as fire, which could cause massive carbon release from the system. Finkral and Evans (2008) found that thinning an over-stocked ponderosa pine forest in Arizona resulted in a net release of 3,114 kg C/ha in aboveground carbon (assuming no storage in wood products). In the event of a stand replacing forest fire, however, this thinned stand is predicted to release 2,410 kg C/ha less than an un-thinned stand experiencing the same intensity fire, although a stand replacing fire is much less likely in the thinned stand.

### **Harvesting**

Harvesting will inherently release some amount of carbon from a forest due to fossil fuels used by vehicles, soil carbon lost through respiration and erosion, and above-ground carbon lost from trees removed from the forest (although this carbon may continue to be stored for long periods of time depending upon whether the wood is being used in long-lived products). There is a growing body of literature, however, on strategies for minimizing carbon loss from forests during harvesting. See Chapter 9, this volume, for a more in-depth discussion of reduced impact logging in the tropics.

#### *Rotation length*

Longer rotations can increase the total carbon stored in a forest as trees continue to add biomass (Paul et al., 2002), and will delay the point at which carbon is released during harvest. However, many A/R projects have the additional objective of producing harvestable timber. Lengthening rotations can cause tension between the dual goals of maximizing timber value and storing additional carbon. Also, lengthening rotations can increase the risk of disturbance, such as fire, if the forest is allowed to become over-stocked (Laclau, 2003b).

#### *Importance of harvest residue*

Whole-tree harvesting has become more common as biofuel markets develop. Leaving residual woody debris in the forest is important for minimizing carbon loss to the system at the time of harvest (Kim et al., 2009) and also helps protect against nutrient leaching and erosion, which helps prevent loss of carbon from the system (Mendham et al., 2003). Stem-only harvesting can produce higher carbon stocks than whole-tree removal harvesting (Jones et al., 2008). Whole-tree removal in turn maintains higher carbon stocking than whole-tree removal that also removes the litter and dead woody debris on the forest floor.

## **MANAGEMENT AND POLICY IMPLICATIONS OF AFFORESTATION/ REFORESTATION (A/R) PROJECTS**

### **Recommendations for land managers**

- Afforestation may result in above-ground carbon additionality, but often

results in adverse impacts on other ecosystem values such as water and soil carbon. Managers should seek to minimize these impacts.

- Reforestation generally results in carbon additionality, and adverse impacts to other ecosystem values are less likely because the land has naturally supported forest in the past.
- Land managers should consider using nitrogen-fixing species to reduce fertilizer inputs and increase biomass production.
- Thinning increases the value of timber and reduces the risk of catastrophic disturbances in a stand, but reduces stand level biomass and carbon. We believe thinning should be used as a risk mitigation strategy for A/R carbon projects, despite the lower carbon stocks that will result.
- There is a growing body of research suggesting that native species plantations are competitive with exotic species from a growth and yield perspective. Land managers should explore opportunities to implement native species silvicultural systems, due to their positive co-benefits.

#### **Recommendations for policy makers**

- Soil carbon is an important component of forest ecosystem carbon stocks. Excluding soil carbon from carbon legislation may result in projects that look additional on paper, but are not additional due to extensive losses of soil carbon that can occur when soil is disturbed, and with changes in hydrology.
- Solely focusing on carbon sequestration, to the exclusion of other forest values (water supply, biodiversity, nutrient depletion), may result in undesired consequences of A/R projects. A “no negative side effects” policy is important for A/R policy.
- Policy makers should consider how to incentivize the international timberland investment community to use native species. Large institutional investors and international companies are responsible for many A/R projects, and they currently do not use mixed species on a large scale.
- While A/R projects are important, primary forests hold even more carbon than A/R forests. Reducing deforestation of primary forests should be a top priority for mitigating climate change through forestry activities.

#### **Recommendations for further investigation**

- More research is still needed into the impacts of A/R projects on water, biodiversity and other ecological values across the many different ecosystems where A/R projects occur.
- While there are a number of studies addressing changes to soil carbon on former crop and pasture land, more research is still needed on the long-term impacts of A/R projects on soil carbon across some of the other ecosystems where A/R projects occur.

- Much less research has been done on managing native species plantations than on managing the major exotic species. More long-term research is still needed to reduce the uncertainty associated with native species plantations.

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## Chapter 12

# The Role of Forest Products in the Global Carbon Cycle: From Forest to Products-in-Use

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### EXECUTIVE SUMMARY

This chapter reviews the role of harvested wood products (HWP) and the forest products industry within the context of global carbon stocks and flows. Harvested wood products can be long term reservoirs of carbon, however, solid wood products, paper, and board manufacturing require large energy and heat inputs, making HWP and carbon a complex topic. A review of published literature revealed the following important considerations:

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The global stock of carbon within forest products is estimated between 4,100 Tg carbon (Han et al., 2007) and 20,000 Tg carbon, with net sink rates estimated between 26 Tg carbon per year to 139 Tg carbon per year. Even assuming the high end of the estimates, this suggests that forest products are still are a minor component of the global carbon budget.

- Manufacturing processes operate on a blend of fossil energy and biomass energy that is a co-product derived from wood waste.
- Newer wood products such as oriented strand board, laminated veneer lumber and I-joists use 80-216% of the energy needed to produce solid sawn lumber. It is unclear whether the lower density of newer wood product materials, given their increased strength and greater utilization of wood resources, offsets the energy intensity per unit of certain newer materials.
- Paper products contain significantly more embedded fossil fuel (carbon) energy than wood products (0.3-0.6 MgC in energy used/MgC for virgin



paper products vs. 0.07 MgC in energy used /MgC for wood products). However, approximately 50% of U.S. paper production is manufactured using recycled paper as a feedstock. Recycled feedstock may reduce or increase GHG emissions relative to virgin pulping depending on the pulping process and energy sources.

- Global transport of wood and paper products accounts for 27% of total fossil carbon emitted within the manufacturing and distribution process.
- Several researchers assert that substitution of wood for other construction materials (e.g., steel and concrete) produces net GHG emissions reductions. These substitution effects may be up to 11 times larger than the total amount of carbon sequestered in forest products annually. Quantification of substitution effects relies on many assumptions about particular counterfactual scenarios, most importantly linkages between increased/decreased forest products consumption and total extent of forestland.

**Keywords:** Harvested Wood Products (HWP), greenhouse gas emissions, round wood, pulp and paper, HWP manufacturing, HWP substitution effects, forest products industry, HWP life cycle analysis.

## INTRODUCTION

Although many policy makers recognize the role forests play in carbon sequestration and climate change mitigation, to date there is no accepted methodology for quantifying and incorporating harvested wood products into global carbon budgets and carbon markets. While there is ample discussion surrounding sustainable forest management and the long-term sequestration of carbon in standing forests, the discussion rarely considers the lifecycle of wood and does not consider the linkages between forest management and end markets for wood products. This chapter reviews the role of the forest products industry and harvested wood products (HWP) within the context of global carbon stocks and flows. It reviews the direct and indirect effects on greenhouse gas (GHG) emissions within the wood products and paper and board sectors. It demonstrates the complexities of including wood products in use by analyzing recent research in life-cycle analysis, manufacturing and use trend data, and literature on the impacts of materials substitution. It concludes by considering areas of further research, such as incorporating the role of forest management in carbon sequestration and further study of carbon benefits currently claimed by proponents of wood product substitution for more energy intensive raw materials.

From the perspective of global carbon stocks and flows, forest products are a heterogeneous group that consists of very short-lived products (e.g., newsprint) to very long-lived products (e.g., furniture, housing stock). Heterogeneity increases further due to different manufacturing processes, energy requirements for production, sources of energy within manufacturing, consumption patterns, end-of-life considerations, and substitution effects.

**Table 1 A framework for evaluating the carbon profile of the forest products industry**

<b>Direct emissions</b>	Manufacturing
<b>Indirect emissions</b>	Transport
	Purchased power
	Landfill CH <sub>4</sub> emissions
<b>Sequestration</b>	Forests
	Products in use
	Products in landfills
<b>Avoided emissions</b>	Combined heat and power applications
	Product recycling
	Substitution effects

Source: Adapted from Miner (2008)

Miner (2008) presents a useful framework for evaluating the carbon profile of the forest products industry (Table 1). In general, direct and indirect emissions represent positive GHG contributions, while sequestration and avoided emissions represent negative GHG emissions relative to a “business-as-usual” scenario (Miner, 2008). The net GHG profile is difficult to quantify because data for several of these processes are imprecise. This is particularly true for sequestration and emissions within solid waste disposal sites (SWDS) as well as substitution effects.

In light of the broad divestment of industrial timberland in the United States (Brown, 1999), largely to timberland investment management owners (TIMOS), it is reasonable to ask whether forest carbon sequestration should be part of the carbon profile of the forest products industry. As many timberland buyers increasingly seek to manage, quantify, and monetize carbon sequestration benefits (Lippke and Perez-Garcia, 2008) alongside traditional timberland management strategies, it is becoming increasingly important to apply a life-cycle analysis to the industry, including the sequestration potential of forestlands as raw material inputs. Similarly, it is also important to quantify products-in-use and in landfill sequestration since these are also key components of the life-cycle of forest carbon in use.

In 2006, the total global annual volume of harvested wood products was 3.42 billion m<sup>3</sup> (FAO, 2007). About 1.65 billion m<sup>3</sup> was industrial roundwood, while 1.77 billion m<sup>3</sup> was fuelwood. Others, however, suggest that harvest was slightly lower (approximately 3 billion m<sup>3</sup> per year), with approximately 1.8 billion m<sup>3</sup> as industrial roundwood, and 1.2 billion m<sup>3</sup> as fuelwood (Nabuurs et al., 2007). These figures may differ on the total fuelwood harvest, since the economic data typically do not include fuelwood. In 2006, developed countries accounted for 70% of total global production and industrial roundwood consumption (FAO, 2007). The largest producers, in order, are the USA, Canada, Russia, Brazil, and China.

Forest products and the forest products industry are unique within the realm of carbon stocks and flows. First, industrial production of forest products typically uses a high proportion of biomass derived from production byproducts as its energy source. Nevertheless, the vast majority of direct fossil CO<sub>2</sub> emissions are still generated in the production phase. Once in use, most forest products do not generate CO<sub>2</sub> emissions; upon disposal, they can generate varying degrees of CO<sub>2</sub> and methane (CH<sub>4</sub>) depending on decomposition rates.

Forest products are often considered to be less energy- and emissions-intensive substitutes for other building materials, particularly concrete, steel, and aluminum (Wilson, 2005; Upton et al., 2008). These substitution effects may play a much greater role in global CO<sub>2</sub> reduction schemes than improvements within the forest products manufacturing process itself (Kauppi and Sedjo, 2001; Miner, 2008). However, while some researchers (e.g., Burschel et al., 1993) regard product substitution as important, they point out that changes in forest management are even more significant. Denman et al. (2007) notes that terrestrial ecosystems, and forests in particular, sequester amounts equal to approximately 25% of total anthropogenic emissions. Thus, the impacts of industry on forestland extent, stocking rates, and land-use conversion must be included in a comprehensive analysis of the carbon footprint of the industry.

The National Council for Air and Stream Improvement (NCASI, 2007) estimates that the net emissions of the forest products industry are largely offset by forest carbon sequestration, although they acknowledge a high degree of uncertainty (Table 2). The largest areas of uncertainty are within transportation-related emissions, forest carbon sequestration, and methane emissions from landfilled forest products. It is important to note that this figure does not take into account any product substitution effects, but does incorporate a large figure for forest carbon sequestration that may not be linked to the production of forest products.

Long-lived wood products-in-use constitute a carbon sink (Skog, 2008), as do some wood products within solid waste disposal sites (SWDS) (Skog and Nicholson, 1998; Micales and Skog, 1997). NCASI (2007) estimates that within the United States the total gross emissions through the forest products value chain in 2005 were 212 Tg CO<sub>2</sub>-equivalent per year, while the forest carbon pool in products (in-use and landfills) grew by 108.5 Tg CO<sub>2</sub>-equivalent per year. In the U.S., landfilled wood products consist of 3% of total carbon stocks within the forest sector, but account for 27% of carbon sequestration (defined as flux in total carbon stocks), which is estimated to average 162 Tg carbon per year (Woodbury et al. 2007). The global stock of carbon within forest products is estimated between 4,100 Tg carbon (Han et al., 2007) and 20,000 Tg carbon (Sampson et al., 1993; IPCC, 1996), with net sink rates estimated between 26 Tg carbon per year (IPCC 1996) to 139 Tg carbon per year (Winjum et al., 1998). Estimates of the total global standing forest carbon stock are 3,590,000 Tg carbon for vegetation only and 11,460,000 Tg carbon for forest biomass and soils (IPCC, 2007). Others estimate that the total carbon stock of the terrestrial biosphere is 3,499,999 Tg C, including non-forest stocks (Fischlin et al., 2007). Even assuming the high estimate of 20,000 Tg carbon for forest products, this suggests that they still are a minor component of the global carbon budget.

**Table 2 Emissions and sequestration estimates for the global forest products value chain**

<b>Value Chain Component</b>	<b>Estimated Net Emissions, Tg CO<sub>2</sub>-eq. per year</b>	<b>Certainty*</b>
Direct Emissions: Manufacturing	262	±20%
Indirect Emissions: Purchased Power	193	±25%
Indirect Emissions: Transport	70	±50%
Indirect Emissions: Landfill-derived Methane	250	-50% to +100%
Net Forest Sequestration	-60	±200%
Sequestration in Forest Products	-540	±50%
Avoided Emissions: Biomass Fuels	-175	±200%
Avoided Emissions: Combined Heat & Power	-95	±200%
Avoided Emissions: Recycling	-150	±200%
Product Substitution Effects	Unknown	N/A

\*Certainty is based on professional judgment as presented in NCASI (2007)

Source: Modified after NCASI, 2007.

However, Woodbury et al. (2007) assert that the forest products sector (including forest growth) provided net carbon sequestration equal to 10% of total U.S. CO<sub>2</sub> emissions in 2005. While forests accounted for 63% of net sequestration, changes in products-in-use and landfilled forest products accounted for 37% of net sequestration, implying that in 2005 the production and disposal of forest products was responsible for sequestering 3.7% of total U.S. CO<sub>2</sub> emissions. NCASI (2007) estimates that in 2005, 52% of gross emissions from the forest products industry was offset by carbon sequestration in products-in-use and products in landfill. It further estimates that annual forest growth on all private lands offset an additional 61% of gross emissions from the forest products industry. USEPA (2008) reports that total U.S. CO<sub>2</sub> emissions in 2005 were 7,130 Tg CO<sub>2</sub>e, which, using figures from NCASI (2007), suggests that forests represent only 1.8% of total U.S. emissions, and that forest products represent only 1.5% of total U.S. emissions.

A potentially broader set of carbon implications arises from the use of wood for energy or other products as a substitute for more carbon-intensive materials. These substitution effects have been explored extensively through comparisons of steel/aluminum/concrete vs. wood housing designs (Wilson, 2005; Perez-Garcia et al., 2005; NCASI, 2007) and use of biomass fuels (Sedjo, 2008). There is less literature, however, on the effects of wood demand on maintaining tracts of forestland. Together, these indirect effects may play a much larger role in GHG reduction than direct effects within the forest products sector.

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USEPA (2008) reports that total U.S. CO<sub>2</sub> emissions in 2005 were 7,130 Tg CO<sub>2</sub>e, which, using figures from NCASI (2007), suggests that forests represent only 1.8% of total U.S. emissions, and that forest products represent only 1.5% of total U.S. emissions.

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## OVERVIEW OF THE GLOBAL FOREST PRODUCTS INDUSTRY

For the purposes of this review, we make a distinction between the forest harvests for land clearing/forest conversion versus the results for the continuous production of forest products such as sawtimber, paper/pulp, biomass, and other forest products. Land clearing (deforestation) is considered a primary driver of anthropogenic CO<sub>2</sub> emissions, accounting for between 17% and 20% of total global CO<sub>2</sub> contributions between 1990 and 2002 (WRI, 2006; Watson et al., 2000). Drivers of deforestation include a variety of sources ranging from fuelwood consumption, illegal logging, and expansion of agricultural land (Stern, 2006) (see Chapters 14 and 16, this volume).

Once harvested, tree boles intended for human utilization are termed “roundwood.” FAO makes a further distinction between *industrial roundwood*, destined for HWP manufacturing, and *woodfuel*, which is roundwood destined for heating, cooking and energy production. The latter, which is categorized as forest-derived biomass, is not analyzed here. While industrial roundwood and paper/paperboard production are concentrated in a few industrialized countries, fuelwood is less concentrated, and more prominent in lesser-developed countries (Table 3).

**Table 3** Production and production concentration of industrial roundwood, paper and paperboard, and wood fuel.

	Industrial Roundwood, 1,000 m <sup>3</sup> per year	Industrial Roundwood, % of Global Production	Paper & Paperboard, 1,000 tons per year	Paper & Paperboard, % of Global Production	Woodfuel, 1,000 m <sup>3</sup> per year	Woodfuel, % of Global Production
Production, Top 10 Countries	1,175,185	71%	263,350	74%	1,061,620	60%
Production, Top 25 Countries	1,445,594	88%	331,510	94%	1,385,578	78%
Production, Total Global	1,645,681	100%	354,490	100%	1,771,978	100%
Top 10 Countries in Production	USA, Canada, Russian Fed., Brazil, China, Sweden, Finland, Germany, Indonesia, France		USA, China, Japan, Canada, Germany, Finland, Sweden, South Korea, France, Italy		India, China, Brazil, Ethiopia, Indonesia, Dem. Rep. of Congo, Nigeria, Russian Fed., USA, Mexico	

Source: Derived from FAO, 2004.

The percentage of roundwood used for woodfuel varies greatly by country. Developed countries typically report low percentages used for fuelwood, while lesser-developed countries generally report a higher proportion of roundwood as fuelwood (Table 4).

## DIRECT EFFECTS OF FOREST PRODUCT HARVEST, MANUFACTURING, AND DISTRIBUTION

### Wood harvesting

Wood products come from trees harvested from natural forests and plantations. The

harvesting process generates significant amounts of by-product, such as branches, leaves and other unmerchantable biomass, which are often either burned or left in the forest to decompose. The proportion of merchantable to unmerchantable biomass varies by forest type, species, and age at harvest. Representative values cited in the literature for North American forests suggest that 20-40% of tree biomass remains in the forest after harvest (Côté et al., 2002; Finkral and Evans, 2008). The variability reflects the diversity of commercially harvested species and forest types, as well as economic factors and harvest technologies.

**Table 4 Roundwood, pulpwood, woodfuel production and production ranking for selected countries and the global HWP industry.**

All values 1,000 m <sup>3</sup> per year	Global Total	Selected Countries								
		US	Finland	Canada	Russia	Japan	Brazil	China	Zambia	Mexico
Industrial Roundwood	1,645,681	414,702	49,281	196,667	134,000	15,615	110,470	95,061	834	6,913
<i>% as Pulpwood</i>	<i>32%</i>	<i>41%</i>	<i>51%</i>	<i>14%</i>	<i>40%</i>	<i>23%</i>	<i>43%</i>	<i>7%</i>	<i>0%</i>	<i>14%</i>
Woodfuel	1,771,978	43,608	4,519	2,901	48,000	114	136,637	191,044	7,219	38,269
<b>Total Roundwood</b>	<b>3,417,660</b>	<b>458,310</b>	<b>53,800</b>	<b>199,568</b>	<b>182,000</b>	<b>15,729</b>	<b>247,107</b>	<b>286,105</b>	<b>8,053</b>	<b>45,182</b>
<i>% of Roundwood as fuelwood</i>	<i>52%</i>	<i>10%</i>	<i>8%</i>	<i>1%</i>	<i>26%</i>	<i>1%</i>	<i>55%</i>	<i>67%</i>	<i>90%</i>	<i>85%</i>
Global Rank of Industrial Roundwood Production		1	7	2	3	18	4	5	78	33

Source: Modified from data in FAO, 2004.

Sustainable management and the use of a formal management plan should be requirements for any forest to be included as a carbon sink under national and international GHG accords. As of 2007, approximately 90% of developed country forests were harvested under sustained yield objectives within a management plan, while only 6% of developing country forests were similarly managed (Nabuurs et al, 2007).

### Wood products manufacturing

Wood products manufacturing uses a majority of all industrial roundwood volume. Its direct manufacturing emissions are a small fraction of total industry emissions, unlike paper and pulp manufacturing, which create much higher direct emissions. Globally, in 2004, 68% of all industrial roundwood volume went to the manufacturing of a variety of solid wood products (FAO, 2007). Using 2004 FAO data, NCASI (2007) estimates that the global solid wood products industry emits 25 Mt of fossil CO<sub>2</sub> per year. Major categories include solid sawn lumber (softwoods and hardwoods), structural panels (plywood, oriented strand board [OSB]), non-structural panels (e.g., particleboard), engineered wood products (laminated veneer lumber, I-joists, glulam), and miscellaneous uses (telephone poles, railroad tracks). Many of these products are manufactured for durable purposes, with the exception of packaging/pallets. Since durable goods usually have product lives of 40-80 years,

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As of 2007, approximately 90% of developed country forests were harvested under sustained yield objectives within a management plan, while only 6% of developing country forests were similarly managed.

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solid wood products have the potential to sequester carbon for significant durations. Furthermore, much of the solid wood stream is then recycled or is deposited in a solid waste disposal site, where it may be sequestered near-permanently (Skog and Nicholson, 1998).

McKeever (2002) estimates that in 1998 the United States, which leads the world in consumption of solid wood products, consumed 0.23 billion m<sup>3</sup> of solid wood products in the following proportions: solid sawn lumber (62%), structural panels (18%), nonstructural panels (12%), engineered wood products (1%) and miscellaneous (8%).<sup>1</sup> Researchers focused on the carbon sequestration potential of the solid wood products sector have offered several conclusions related to product mix and manufacturing:

<sup>1</sup> Data include domestic and imported products.

1. Since 1970, the rate of resource utilization (the percentage of roundwood that ends up in final product form) of the U.S. solid wood products industry has increased significantly, despite a recognized reduction in size and quality of roundwood inputs. Yields from raw materials have increased, and inputs of petroleum-based additives in engineered and panel products have decreased.
2. Since 1970, the product mix within the solid wood products industry has shifted from solid sawn lumber and plywood to a mixture of engineered wood products and OSB. This is likely due to changes in quality of roundwood inputs and demand for uniform, high-performance engineered products (Meil et al., 2007).
3. The industry produces a substantial amount of its energy needs through biomass electricity and heat production, which are often adjacent to manufacturing facilities.

### **Carbon management implications of trends in solid wood product manufacturing**

Within the wood products sector, there is a strong trend toward engineered products such as glue-laminated lumber, I-joists, and non-plywood structural panels such as oriented strand board (OSB). Proponents tout these products for their load-bearing strength and uniformity relative to solid sawn wood (Meil et al., 2007). They can also be manufactured from small-diameter roundwood and/or scraps from other processes. Because these products have been allowed under the two major international building codes (IBC/IRC), are favored by builders for their uniformity and strength, and allow for greater economic utilization of harvested fiber, it is unsurprising that this is the fastest growing sub-segment of the solid wood products industry (Meil et al., 2007). By volume, these products made up about 1% of total U.S. roundwood consumption as of 1998. Sales growth of engineered wood products increased 30.2% over 2000-2004 (McKeever, 2002). In contrast, the American Plywood Association projects that solid sawn lumber consumption will drop below 4 billion ft<sup>3</sup> in 2012, implying no growth in volume between 1998 and 2012.

Because solid wood manufacturing encompasses a mix of solid sawn wood and engineered wood products, it is worthwhile to examine the carbon footprint of each

major segment. A series of studies by Wilson et al. (2005) and Kline (2005) conducted as part of the CORRIM II study<sup>2</sup> provides carbon and energy consumption data for the production of various solid wood products (Table 5). CO<sub>2</sub> emissions by product type range from 202 kg CO<sub>2</sub>/m<sup>3</sup> to 672 kg CO<sub>2</sub>/m<sup>3</sup>, with U.S. Southern OSB production resulting in the highest emissions by volume<sup>3</sup> (Puettmann and Wilson, 2005). Variability within a product type arises from differing regional energy sources and year of analysis. Solid sawn lumber production is not substantially lower in CO<sub>2</sub> emission/volume than the engineered wood products. However, Pacific Northwest plywood generated 24% lower CO<sub>2</sub> emissions than solid sawn wood.

**Table 5 Carbon dioxide (CO<sub>2</sub>) emissions in the cradle-to-gate life cycle of a wood building product from the generation of the forest through product manufacturing** Note: I joists are made of OSB and LVL, and could not be included in this table because of conversion problems. Glulam = glue laminated timber beams; LVL = laminated veneer lumber, OSB = oriented strand board.

Product	Pacific Northwest Production				Southeast Production				
	Glulam	Lumber	LVL	Plywood	Glulam	Lumber	LVL	Plywood	OSB
CO <sub>2</sub> emissions (biomass), kg/m <sup>3</sup>	230	160	141	146	231	248	196	229	378
CO <sub>2</sub> emissions (fossil), kg/m <sup>3</sup>	126	92	87	56	199	62	170	128	294
CO <sub>2</sub> emissions, total, kg/m <sup>3</sup>	356	252	228	202	430	310	366	357	672
CO <sub>2</sub> emissions (biomass), kg/m <sup>3</sup>	65%	63%	62%	72%	54%	80%	54%	64%	56%
CO <sub>2</sub> emissions (fossil), kg/m <sup>3</sup>	35%	37%	38%	28%	46%	20%	46%	36%	44%
Total energy, MJ/m <sup>3</sup>	5,367	3,705	4,684	3,638	6,244	3,492	6,156	5,649	11,145
Product yield, log to product		53%		51%		41%		50%	71%
Product yield, other wood inputs to product	82%		N/A		82%		N/A		
Description of other wood inputs	Dry, planed lumber		Veneer		Dry, planed lumber		Veneer		

Source: Derived from Puettmann and Wilson (2005).

Most engineered wood products contain (by mass) 5-15% in additives such as petroleum-based adhesives, waxes, and resins. These are created under more intensive manufacturing processes. Because these products are stronger, less wood fiber is required within the construction process relative to solid-sawn lumber. For example, I-joists use approximately 62-65% of the wood fiber of a solid joist, but their production is more energy intensive. As a result, substitution of I-joists for solid-sawn lumber provides negligible opportunities for CO<sub>2</sub> emissions reduction (Perez-Garcia et al., 2005). Moreover, substitution of OSB for plywood reduces total carbon emissions only by 3-4%.

Resource utilization studies conducted in 1976 and again in this decade (Meil et al., 2007) document increased utilization of by-products and co-products while providing interesting data on product yields from raw materials. In 1970, the softwood lumber industry had a 35% utilization rate by weight (e.g., conversion of

<sup>2</sup> From [www.corrим.org](http://www.corrим.org) (accessed 12/30/08): "The Consortium for Research on Renewable Industrial Materials (CORRIM) has been organized to update and expand a 1976 report by the National Academy of Science regarding the impacts of producing and using renewable materials." CORRIM researchers have produced life-cycle analyses for major solid wood products include dimension lumber, OSB, LVLs, and other engineered products.

<sup>3</sup> Meil et al. (2007) assert that this is largely due to the use of regenerative thermal oxidizer (RTO) units which are a critical element of air emissions control in OSB manufacturing.



raw logs into the primary product). This rose to 45% in 2000. The authors note that this efficiency improvement occurred within the context of decreasing roundwood quality during the period. Efficiency gains for softwood plywood showed a 7% improvement. However, a much greater proportion of plywood byproducts were used as raw materials for other products, such as nonstructural panels, rather than being burned or landfilled. In addition, over the same period, adhesive and resin content in plywood was reduced by 17% (Meil et al., 2007). The authors therefore assert a reduction of 62.7 kg of fossil-derived CO<sub>2</sub>/m<sup>3</sup> of softwood lumber produced in 2000 relative to 1970.

Biomass has been an important, carbon-neutral energy source for the forest products industry. Biomass is considered by Watson et al. (2000) and others to be a “carbon-neutral energy source” because it does not generate fossil carbon. Within the forestry sector, forest regeneration is thought to offset volatilized carbon from biomass energy production. Yet, within the IPCC framework, changes in forest regeneration are reported separately as land-use change (Watson et al., 2000; IPCC, 2007), which makes it difficult for forest products industry book-keeping to include records of the life cycle of their products for the purposes of calculating carbon stocks. Over the past thirty years, the industry has improved utilization efficiency for materials by creating value from products once burned for energy, and by burning for energy products that were typically burned solely for disposal purposes (Wilson, 2005). Historically, the industry burned bark and other “wet” residues in uncontrolled outdoor burners variously termed “teepee” or “beehive” burners, with significant particulate emissions and zero energy recovery. Only sawdust and planer shavings were converted into energy due to the cost and conversion efficiency of boilers. Today, in developed nations, it is more common for all residue, including bark, mill-ends, sawdust and shavings to be burned for the cogeneration of heat and electric power. These outputs are used to drive manufacturing processes within modern solid wood product mills. But they often remain uncounted in carbon budgeting.

Two studies from the 1970s indicate that historic energy recovery was low. Grantham and Howard (1970) indicate that 25% of residual byproducts were used as fuel, and another 37% transferred to other facilities as raw materials. Corder et al. (1972) claim that 26% (for lumber) and 24% (for plywood) of byproducts were used for fuel. Between 1970 and 2000, bark and “wet residues” began to be used as fuel for combined heat and power applications at manufacturing sites. As these trends continued, byproducts traditionally used for energy production, such as sawdust and planer shavings, began to be sold as co-products.

### **Pulp and paper manufacturing**

In 2004, the pulp and paper industry consumed approximately 32% of all industrial roundwood produced globally (FAO, 2007). NCASI (2007) estimates that pulp and paper manufacturing processes globally emit 195-205 Mt of fossil CO<sub>2</sub> per year (compared to 25 Mt CO<sub>2</sub> per year for solid wood products). The pulp and paper industry generally produces products that are shorter-lived than the solid wood products segment, ranging from various grades of newsprint and paper to

paperboard (see Chapter 13, this volume). The production of paper products from virgin fiber is considerably more energy intensive than all solid wood products, since wood fiber must be converted (chemically or mechanically) from a mixture of cellulose, hemicellulose, and lignins into a cellulose-dominated pulp for papermaking. In 1998, the paper manufacturing industry ranked as the United States' fourth largest emitter of greenhouse gases, following petroleum, basic chemicals, and metals (EIA, 2006). Using 1991 data, Subak and Craighill (1999) estimate that the paper and pulp industry directly and indirectly accounted for 1.3% of total global fossil carbon emissions in 1993.

Industry segments vary in production volumes, carbon intensity, manufacturing processes, and estimated service life. In 2006, the United States produced 41.8 million short tons of paper and 50.4 million short tons of board products. In contrast to flat or slowly growing markets for solid wood products (McKeever, 2002), the U.S. paper and board markets in total have been declining since 1999 (Irland, 2008), likely the result of a transition away from newsprint consumption. Furthermore, production has dropped faster than consumption as significant industry segments have moved offshore (e.g., China now dominates global packaging markets) (FAO, 2007).

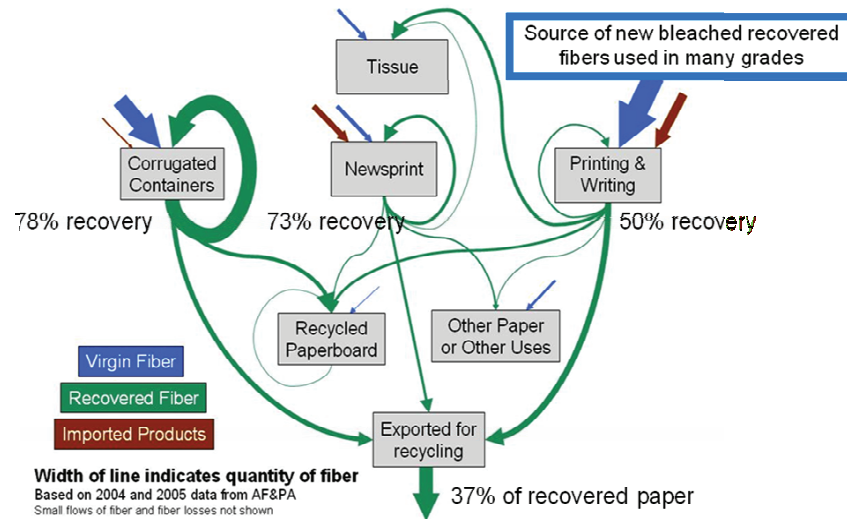
International trade in pulp, paper, and board products is considerably more developed than trade in raw sawtimber and solid wood products. A different set of nations is dominant within global production of paper and paperboard products (Table 3). Additionally, recycled fiber streams play a much greater role in paper manufacturing relative to solid wood products manufacturing (Falk and McKeever, 2004).

Paper industry inputs vary by product type, and include (i) industrial roundwood, (ii) chips as a co-product of solid wood product manufacturing and (iii) recycled fiber. Certain products require more virgin fiber for tensile strength, while other products can be produced with predominantly recycled fiber. Miner (2008) documented a complex fiber supply web within the industry (Figure 1). Of the 100 million tons of paper consumed annually within the U.S. approximately 53.4 million are recovered for recycling. A 2008 press report from Forestweb (Irland 2008) indicates that paper manufactured from 100% recycled pulp results in 1,791 kg/ton of CO<sub>2</sub> emissions, vs. 4,245 kg/ton of CO<sub>2</sub> emissions from paper manufactured from virgin pulp. However, there is some debate over the role of recycled fiber in reducing GHG emissions within the industry. The de-inking and recycling process is energy intensive, and typically involves 100% purchased power (vs. in-house biofuel-derived power in virgin pulp manufacturing). Some researchers suggest that the climate benefits of recycled material arise from the avoided CH<sub>4</sub> emissions from decomposing paper within landfills (Subak and Craighill, 1999; NCASI, 2007).

Using Finland's forest products industry, Pingoud and Lehtilä (2002) estimate that across pulping processes and fiber sources, the proportion of fossil-based carbon emissions per wood-based carbon in end products (Mg carbon/Mg carbon) is 0.07 for sawn wood and 0.3-0.6 for paper in the manufacturing stage, suggesting that paper is 428% to 857% more fossil carbon intensive than sawn wood by mass. They also found that direct fuel, heat, and electricity demands for the production of 11

grades of pulp in Finland in 1995 can dramatically vary (Pingoud and Lehtilä, 2002) (Table 6).

**Figure 1** The complex supply web of the forest products industry



Source: Miner, R., 2008. The Carbon Footprint of Forest Products. Presentation at Yale School of Forestry & Environmental Studies; data from American Forest & Paper Association. Reprinted with permission.

Chemical pulping uses either a kraft (sulfate) process or sulfite process to dissolve lignins, which are burned with other derivatives to recover pulping chemicals and to provide process heat (Côté et al., 2002). This process leaves cellulose fibers largely intact for high-quality papermaking. Mechanical pulping uses fiber more efficiently, yielding a lesser amount for biofuel as a process energy, and increasing the need for purchased electricity (Pingoud and Lehtilä, 2002). Chemical processes result in 50-55% loss of fiber by weight, while recovery of recycled paper results in a 16-18% loss of fiber by weight. Fiber that does not end up in the final product is generally burned in the production process or landfilled (Côté et al., 2002). Industry-wide, 56% of all energy needs are met with biofuel co-products (Davidsdottir and Ruth, 2004). Farahani et al. (2004) have highlighted a new technology, black liquor gasification-combined cycle (BLGCC), which has the potential, under certain conditions, to fully offset energy usage within the chemical pulping process. In this case, using less recycled feedstock actually improves the GHG emissions profile by providing greater opportunities to use biomass and black liquor as energy feedstock.

In general, mechanical pulping is less energy intensive, although it also uses a greater proportion of purchased electricity in its manufacture. Given the reputation for energy and process efficiency of the Nordic paper and pulp industry (Subak and Craighill, 1999), these figures may not be globally representative, yet are among the few data points available on this topic.

**Table 6 Energy inputs and ratio of embedded carbon in raw material vs. final product under a variety of pulping processes in Finland**

	<b>Total Production, Gg per year</b>	<b>Direct Fuels, MWh/Mg</b>	<b>Heat, MWh/Mg</b>	<b>Electricity, MWh/Mg</b>	<b>C in raw material/ C in final product</b>
<b>Mechanical</b>					
GWP, B	801		0	1.55	1.2
GWP, NB	1167		0	2.1	1.23
TMP, NB	923		-0.75	2.4	1.2
TMP, B	801		-1.17	3.37	1.24
CTMP	105		0.56	1.65	1.25
SCP	509		1.06	0.4	1.45
<b>Chemical</b>					
HSUP, B	2174	0.39	3.07	0.69	2.46
SSUP, NB	680	0.52	2.77	0.57	2.56
SSUP, B	2928	0.52	3.33	0.75	2.71
<b>Recycled</b>					
REC, NB	180	0	0	0.1	1.07
REC, B	272	0.25	0.17	0.4	1.17
<b>Total</b>	<b>10540</b>				

\* In Finland, 51% of produced chemical pulp was dried in 1995 (Carlson and Heikkinen 1998). This is included in the energy demand figures.

Abbreviations used: GWP = groundwood pulp, TMP = thermo-mechanical pulp, CTMP = chemi-thermo-mechanical pulp, SCP = semi-chemical pulp, HSUP = hardwood sulphate pulp, SSUP = softwood sulphate pulp, REC = recycled pulp, B = bleached, NB = unbleached.

Source: From Pingoud, K., Lehtila, A., 2002. Fossil carbon emissions associated with carbon flows of wood products. Mitigation and Adaptation Strategies for Global Change 7, 63–83. Reprinted with permission.

Similar to trends within the solid wood products industry, the paper and pulp sector has experienced process, energy efficiency, and resource utilization improvements since 1970. IEA (1993) documents a 0.8% decrease in energy intensity of OECD-country paper and pulp making processes from 1968 to 1990. Nevertheless, as of 2002, the paper and pulp industry remains the second highest manufacturing sector on an energy intensity basis (with petroleum/coal as the highest) (Davidson and Ruth, 2004). This estimate does not take into account the relatively high proportion of energy derived from biomass fuels within the forest products industry, approximately 40% in the United States in 1998 (EIA, 2008).

### **Carbon implications of transport and international trade of forest products**

Transportation of forest products, both as raw industrial roundwood and as consumer products, has been recognized as a significant potential source of fossil

carbon emissions (Pingoud and Lehtilä, 2002; NCASI, 2007). Research indicates that some forest products can travel large distances prior to and following manufacture, via overland freight or cargo ship. Globally, NCASI (2007) estimates that product transport results in fossil carbon emissions of approximately 70 million tons CO<sub>2</sub> per year, or approximately 27% of total fossil carbon emitted within manufacturing and distribution processes. Pingoud and Lehtilä (2002) examined transportation related emissions in Finland, documenting a wide range of transportation modes and distances. Their research concluded that transportation from harvest site to mill, and from mill to consumer, accounted for 22% and 20% respectively of total fossil carbon emitted within manufacturing and distribution processes.

### Indirect effects of the forest products industry on carbon emissions

As noted above, the forest products industry's contribution to total global GHG emissions is minor, despite its high energy intensity (NCASI, 2007), partly due to its significant use of biomass fuels to power manufacturing processes, and its long-lived products, which sequester carbon in products-in-use and landfilled products. Beyond purchased power, transportation, and landfill methane emissions related to forest products, the forest products industry offers products that may be less fossil carbon-intensive than substitute materials such as concrete, aluminum, and steel. To the extent that increased use of forest products results in an expansion of timberlands operated on a sustained-yield basis, substitution effects may have a greater impact on net carbon sequestration beyond a comparison of the embedded energy within various substitutable building materials.

Gustavsson et al. (2006) describe four GHG emissions-related aspects to materials substitution: (i) emissions from fossil fuel use over the life cycle of the product (e.g., production, transportation, end use and waste management); (ii) replacement of fossil fuels with biomass energy within the production phase; (iii) carbon stock changes in forests, products-in-use and landfilled materials; and (iv) GHG emissions from industrial process reactions in such areas as cement and steel production. While it is impossible to accurately quantify all actual and counterfactual outcomes within this framework, Kauppi and Sedjo (2001) indicate that the range of possible substitution effects may be up to 11 times larger than the total amount of carbon sequestered in forest products annually. This suggests that minor changes in consumer preference for materials can have a big impact on the overall GHG emissions profile of the construction sector.

In many applications, forest products are interchangeable with rival products, typically plastics, metals or concrete (Upton et al., 2008). Researchers have compared the carbon footprint of forest products relative to some of these materials, and concluded that increased use of forest products within the construction sector would result in decreased GHG emissions (Wilson, 2005; Upton et al., 2008). Currently, in the U.S., wood framing techniques are used in approximately 90% of new housing starts (Upton et al., 2008). This percentage is much lower in other regions of the world, particularly outside of North America and Northern Europe (Gustavsson et al., 2006).

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In the lifetime of a house, there are two primary sources of carbon emissions: the construction of the structure, and the energy requirements to heat and cool the structure over its lifetime. It is difficult to compare wood vs. other building materials because alternative materials have different thermal characteristics. For example, the thermal mass associated with concrete buildings may reduce heating and cooling costs, thereby lowering carbon emissions during building operation (Upton et al., 2008).

Upton et al. (2008) project that wood-framed single-family houses require 15-16% less total energy and emit 20-50% less fossil CO<sub>2</sub> to build than non-wood houses made of steel framing products. This conclusion relies on several key assumptions about the ratio of embedded energy in housing relative to energy expended to heat and cool the house over its lifetime, as well as assumptions regarding the fate of forests used or not used for the production of industrial roundwood (Upton et al., 2008). Wilson (2005) found that the wood-framed house had a Global Warming Potential Index (a measure of total GHG emissions, not energy usage, as in Upton et al. (2008)) 26% and 31% lower, respectively, than model steel and concrete house designs. These figures represent only the embedded energy within the production of the house, not its operation. These figures are supported by Gustavsson and Sathre (2006), who conducted a sensitivity analysis around uncertainties and variability within the production of both concrete and wood. Using plausible inputs, wood building materials had lower energy costs relative to concrete in all cases analyzed.

Perez-Garcia et al. (2005) characterize the substitution effects throughout the value chain from forest to landfilled product. This analysis demonstrates that forests will accumulate carbon in the absence of harvest or disturbance, but concludes that shorter, more intensive forest management results in a greater amount of carbon sequestration because a carbon pool accumulates within housing stock, where it is sequestered for a long period. Furthermore, with substitution effects, the use of wood products offsets concrete or metal construction, providing a greater benefit than either the forest carbon pool or the forest product carbon pool. In short, intensive forest practices create a “positive carbon leakage” through greater use of wood products in the market place.

Several studies examining substitution posit that greater use of forest products will result in greater retention of working forestlands, or conversely, that less use of forest products will hasten conversion of working forestlands to other land uses (Wilson, 2005, Perez-Garcia et al., 2005; Upton et al., 2008). Regardless of the validity of this assumption, it is important to recognize that each author implicitly or explicitly recognizes that carbon fluxes within forestlands are several orders of magnitude greater than any identified substitution effect. Thus, it is worth examining how and whether the forest products industry has any effect on the extent and condition of forestlands relative to other factors.

### **SIGNIFICANT FACTORS IN ASSESSING THE EMISSIONS PROFILE OF THE FOREST PRODUCTS INDUSTRY**

The emissions profile of the forest products industry is more complex than many other industries, with GHG sources and sinks found within product manufacturing

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and end-of-life considerations. Furthermore, product and energy substitution effects and decisions surrounding forest management result in GHG sources and sinks that are several orders of magnitude greater than those directly attributed to industrial processes. Within product manufacturing, the key factors appear to be (i) selection of energy source, (ii) type of pulping process, and (iii) emissions and pollution control requirements. In particular, lumber mills are faced with multiple markets for co-products ranging from chips to sawdust to biomass energy feedstocks. As markets for chips and sawdust grow, it may be more economical to rely on purchased power for mill operations. This will have a significant impact on the carbon footprint of the industry.

## CONCLUSIONS AND MANAGEMENT AND POLICY IMPLICATIONS

As policymakers focus on the role of forests and HWP in mitigating climate change, additional research is needed to fully understand the relationships among climate policy, the forest products industry, consumers, and forests. Management and policy implications are summarized under the topics forestlands, substitution, and policy initiatives.

### Forestlands

- The potential to sequester carbon in forests is much larger than the potential to sequester carbon in forest products. Minor changes in forest extent have much greater impacts on GHG emissions than the forest products industry. Some researchers (Kauppi and Sedjo, 2001; NCASI, 2007) refer to the beneficial role that the forest products industry plays in maintaining sustained-yield forestland.

### Substitution

- Each major building materials industry (wood, steel, and concrete) has published studies suggesting that their products are superior from the perspective of climate change mitigation. Which conclusions are most supportable? Given that climate considerations are currently an externality, what factors drive materials selection? Will a carbon price signal be sufficient to overcome these factors?

### Policy initiatives

- The rise of biomass energy use in the forest products industry as well as increasing utilization of wood products have largely been driven by the competitive nature of the industry as well as the need to lower costs while seeking new sources of revenues, particularly for by-products and co-products that had historically not generated an economic return to the industry. In many ways, these have been economic trends that have had a net carbon benefit. Under certain economic conditions, however, forest products

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In short, intensive forest practices create a “positive carbon leakage” through greater use of wood products in the market place.

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manufacturers may be inclined to alter the manufacturing process, which could result in incremental emitting activities under certain scenarios, particularly if it lowers costs for a profit maximizing entity.

- To date, policymakers have not fully considered the role harvested wood products can play in climate change mitigation and have not linked forest management practices to the full life cycle of harvested wood products. Incentives should be considered to support the use of recycled materials, to encourage such activities as product substitutions, certification systems in construction (such as LEED), industrial energy efficiency, and to encourage biomass fuel sources.
- There are many factors that will favor or disfavor wood as a construction material or energy source. These include relative price, technology, economic growth, policy, market efficiency, socioeconomic factors, and quality and quantity of energy and materials (Gustavsson et al., 2006). Recognizing that wood products are still largely a cyclical industry driven by global GDP, such policies could begin to introduce longer-term, secular demand for wood products that encourage investment in wood that is both economically and environmentally sound.

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## Chapter 13

# The Role of Forest Products in the Global Carbon Cycle: From In-Use to End-of-Life

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### EXECUTIVE SUMMARY

Climate change mitigation policy has focused attention on the protection and enhancement of carbon sinks. The global stock of harvested wood products (HWPs) has garnered specific attention as roughly 1.6 billion m<sup>3</sup> of industrial roundwood is extracted yearly with the purpose of producing both long- and short-lived wood products. This chapter examines HWPs, their end-of-use pathways and the role of HWPs in the global carbon cycle. A review of published literature on this topic revealed the following important considerations:

- Studies have shown that total stock of HWPs in use is increasing each year. Further, emission reductions are achieved as the production of HWPs in many cases is fueled primarily with energy from burning wood byproducts, thus displacing fossil fuel emissions and achieving greater reductions.
- The end-of-use pathways of HWPs are equally promising. Once discarded, HWPs can be burned for energy production, recycled or reused, or put in landfills, where the carbon remains indefinitely due to anaerobic conditions. However, HWPs discarded in landfills create methane, a greenhouse gas that is 24 times more potent than CO<sub>2</sub>, thus potentially offsetting gains from carbon storage.
- The methods and assumptions used to estimate the role of HWPs in the global carbon cycle vary, resulting in a wide range of data. Potential yearly increase to the global HWP carbon stock varies with estimates ranging from 26 to 139 Tg C.

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- It is recommended that end-of-use pathways be incorporated into HWP carbon stock calculation models, as failure to do so would provide estimates with a high degree of error.

**Keywords:** Harvested Wood Products, Green House Gas Inventories, Lifecycle Analysis, Landfill Gas

## INTRODUCTION

Understanding the role of harvested wood products (HWP) in the global carbon cycle is essential if appropriate policy promoting the greater use of HWPs is to be implemented on a national or even international level under multi-lateral agreements in a post-Kyoto protocol regime (Rueter, 2008). Studies that quantify current global stocks of HWPs vary greatly, as calculation methods are dependent on critical assumptions regarding product life, decay rates, and system boundaries (Pingoud et al., 2003; Green et al., 2006). A lack of data on the usage and disposal of HWPs adds to the difficulty of quantifying this global carbon stock (Kuchli, 2008). Opinion on system boundaries is divided across the literature. The topic of landfills is a major part of this debate as models that include “end-of-use” within their system boundaries are intrinsically tied to assumptions made regarding the level of methane (CH<sub>4</sub>) capture from landfills. Landfill material makeup has a significant impact on the magnitude of CH<sub>4</sub> production, while the landfill design greatly influences the ability to capture landfill gases and convert methane passively through oxidation.

This chapter reviews the literature on the carbon stock of HWPs and outlines the currently accepted research on the topic of product life spans and HWPs in landfills. It also discusses the end-of-use pathways of HWPs and their carbon implications.

### Definition of harvested wood products (HWPs) and related terms

*Harvested wood products* (HWPs) can be defined as wood-based materials that, following harvest, are transformed into commodities such as furniture, plywood, paper and paper-like products (Green et al., 2006). The term HWP is further simplified by the International Panel on Climate Change (IPCC) defining it as all wood material (including bark) that is transported off harvest sites. It does not include woody biomass, commonly referred to as slash or residual material, left at harvest sites (Pingoud et al., 2003).

*Roundwood* refers to the logs which are extracted during a timber harvest. The FAO defines roundwood as wood in its natural state after it has been harvested, including logs that have undergone minimal transformation and may be without bark, rounded, split, or roughly squared. Roundwood is used as either woodfuel or industrial roundwood, which is used to produce HWPs.

HWPs are categorized into two groups: *solid wood products* (SWPs) and *paper products*. Solid wood products consist of sawn wood and wood-based panels, typically measured in cubic meters. Paper products are defined as paper and

paperboard which are measured in dry tons (Green et al., 2006). In many cases, HWP are further transformed into different product classes and categories throughout their lifecycle due to recycling (Pingoud et al., 2003).

### ESTIMATE OF CARBON IN HWPs

Global estimations of yearly HWP production are derived from statistics collected by FAO on the production of roundwood. In the United States, the USDA Forest Service keeps statistics on roundwood harvests and HWP production based on data collected from government agencies and industry. The FAO reports that, globally, 1.65 billion m<sup>3</sup> of roundwood is extracted annually for HWP production (FAO 2007). The United States produces approximately 425 million m<sup>3</sup>, or 25%, of global roundwood intended for HWPs (Howard, 2006). If we were to convert this global roundwood production figure to carbon, it would be very large.

However, production losses occur as roundwood is processed into different products, and assumptions on the magnitude of these losses greatly influence the final calculations. First, it is assumed that roughly 50% of harvested roundwood logs is lost as residues (Gardner et al., 2004), which brings the total to 825 million m<sup>3</sup>. Data from 2004 show that the paper products industry consumed 32% of the total roundwood production, which would account for 264 million m<sup>3</sup>, while solid wood products accounted for 561 million m<sup>3</sup>. However, these figures are further reduced when losses from final product finishing are taken into account. Skog and Nicholson (2000) assumed an 8% loss for solid wood products, and 5% for paper products, during finishing. This would mean that 516 million m<sup>3</sup> of solid wood products and 251 m<sup>3</sup> of paper products comprises the total annual global production of HWPs. Using the same assumptions on production losses, United States yearly production of HWPs would amount to 133 million m<sup>3</sup> of solid wood products (SWP) and 65 million m<sup>3</sup> of paper products (see Table 1).

Estimates of the total carbon sequestered in HWPs globally vary widely from 4,200 Tg<sup>1</sup> C (IPCC, 2000) to 25,000 Tg C (Matthews et al, 1996). Similarly, estimates of the net annual sink from HWPs ranges from 26 Tg C/yr to 139 Tg C/yr in these same reports.

Compared to the 38,000 Tg CO<sub>2</sub>e in estimated worldwide emissions in 2004 (IPCC, 2007), which equates to 139,300 Tg C, the total amount of carbon sequestered annually in HWPs is small. There are several reasons to explain the wide range in these figures on HWP annual sink. First, estimates will vary based on the assumptions made about average production losses and wood densities. The choice of wood density can have considerable impact on the results (Stern, 2008). Secondly, HWP stock estimates frequently do not distinguish between HWPs in use versus those in landfill waste (Pingoud et al., 2003). A standard methodology for converting HWP mass into carbon equivalents is necessary to compare data reported from different countries along with better estimates for country specific trends in landfill waste.

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<sup>1</sup>Tg (Tetagram) = 10<sup>12</sup> g

**Table 1 Global production of HWPs in 2000 according to FAOSTAT 2002.** The associated carbon fluxes have been estimated by assuming approximately that the dry weight of coniferous wood would be 0.4 t/m<sup>3</sup> and non-coniferous 0.5 t/m<sup>3</sup> and that the carbon fraction in biomass is 0.5. In addition, the estimated charcoal production was 0.04 billion t/yr (metric tons per year). The production of wood residues was 0.06 billion m<sup>3</sup>/yr and chips and particles 0.16 billion m<sup>3</sup>/yr, these being mainly by-products of wood processing.

<b>PRIMARY PRODUCTS</b>		
	billion m <sup>3</sup> /yr	Pg C/yr
<b>Roundwood</b>	<b>3.1</b>	<b>0.71</b>
<b>Wood Fuel</b>	<b>1.5</b>	<b>0.37</b>
<b>Industrial Roundwood</b>	<b>1.6</b>	<b>0.34</b>
Pulpwood (Round & Split)	0.48	0.11
Sawlogs + Veneer Logs	0.95	0.20
Other Indust Roundwd	0.15	0.03
<b>SEMI-FINISHED PRODUCTS</b>		
	billion m <sup>3</sup> /yr	Pg C/yr
<b>Sawnwood</b>	<b>0.42</b>	<b>0.09</b>
<b>Wb-Panels + Fibreboard</b>	<b>0.22</b>	
	billion t/yr	
<b>Paper + Paperboard</b>	<b>0.32</b>	<b>0.15</b>

Source: Pingoud, K., Soimakallio, S., Perala, A.L., Pussinen, A., 2003. Greenhouse gas impacts of harvested wood products: Evaluation and development of methods. VTT Technical Research Center of Finland, Research Notes 2189. Reprinted with permission.

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The carbon embodied in short-lived products can be released quickly back into the atmosphere after rapid decomposition, while long-lived products can store carbon for many years.

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### **CALCULATING USEFUL LIFETIMES OF HWPs**

The figures above give us a rough estimate of the potential yearly input to the global carbon stock of HWPs. However, since these calculations fail to recognize the finite life of HWPs, even these rough estimates are inflated. Lifespans of HWPs vary significantly by product type and must be accounted for accordingly. The carbon embodied in short-lived products can be released quickly back into the atmosphere after rapid decomposition, while long-lived products can store carbon for many years. Some wood or paper items such as antiquities and historic buildings are expected to have very long lives (in excess of 100 years) (Skog et al, 1998). However the majority of paper products have a high rate of retirement, lasting only weeks (Marland and Marland, 2003).

The lifespan attributed to products has a major impact on the outcome of estimates on the stock of HWPs. Although it is critical to determine the average lifespans of various HWPs, it is difficult due to a lack of data on product use and disposal (Stern, 2008). In response, some believe that HWP lifespans should not be viewed as empirical but as parameter values used in models (Pingoud et al., 2003). Data on HWP use has shown that rate of retirement of HWPs from end uses is more or less constant for a period, then accelerates for a while near the median life, and finally slows down after the median life (Skog and Nicholson, 2000). As a result, the average lifespan of HWPs is much shorter than in ideal conditions (Pingoud et al., 2003). Because of the difficulty of determining lifespans, it is common to see conflicting values for the lifespan of the

same products in different studies. For example, a review of studies has shown that the average lifespan of pallets ranges from 2-20 years (Pingoud et al., 2003).

Data on average lifespan is then used to model how HWPs are discarded and ultimately oxidized. Much of the literature uses the term decay to describe both the retirement of products from use as well as the decomposition of products (Dias et al., 2009). The decay parameters for products in use are different from those out of use (in landfills, for example) where decay of HWPs may be halted almost completely. Most studies, however, do not model the decay of HWPs that are out of use (Pingoud et al., 2003). Instead these studies model the retirement of HWPs and assume that oxidation occurs at different rates as a function of the product's retirement function.

The type of decay model used has a significant impact on estimating the HWP carbon stock as it determines the timing of carbon releases through oxidation. Since HWP carbon stock represents (in effect) a postponement of future carbon emissions, the timing of their release becomes a critical assumption in estimating current HWP carbon stock increases and magnitudes. Numerous methods for modeling the carbon release of HWPs exist, each with varying effects on HWP carbon stock estimates (Dias et al., 2009).

The first method of modeling HWP oxidation is to assign an exponential decay rate to a product. This is often done by assigning each end use HWP a carbon half-life which represents the time in which half of the carbon embodied in the end-use product is no longer there and has been emitted back into the atmosphere. This exponential decay model assumes that 90% of the carbon in HWPs is released in 3.3 times the assigned half life. Under this model, carbon release begins immediately once a product is in use and occurs at a greater rate earlier on in the life of the product and slows as the product ages. The second approach assumes that products of this type all have the same age, which is set to the product's average lifespan. In the model, 100% of the carbon remains embodied in the HWP until it is discarded, at which time all the carbon in the HWP is then released into the atmosphere. The third method follows a linear model in which a percentage of the initial amount of carbon in the HWP is discarded each year. The year in which all the carbon has been released is the maximum lifespan of the HWP type. Half of the time needed to reach the maximum lifespan is the product's average lifespan. The emissions profile of these models can be linear, exponential or equal (Figure 1) (Skog and Nicholson, 2000).

The different methods in modeling carbon release from HWPs clearly show how assumptions concerning product lifespan can significantly alter estimates. The most rudimentary model is that which assumes products of a certain type have an equal age and release 100% of their carbon at the time of retirement. This model does not account for carbon that is released into the atmosphere from products that are discarded before reaching their average lifespan. This method may mask carbon emissions that are occurring from HWP use and may inflate estimates of the annual increase in the HWP carbon stock. The linear and exponential decay functions both have carbon emissions occurring from the start of a product's life, which accounts for products that are discarded much earlier than reaching the average lifespan. The exponential function creates a scenario where carbon emissions occur much faster in the beginning and slow as a product gets closer to reaching its average lifespan. HWP

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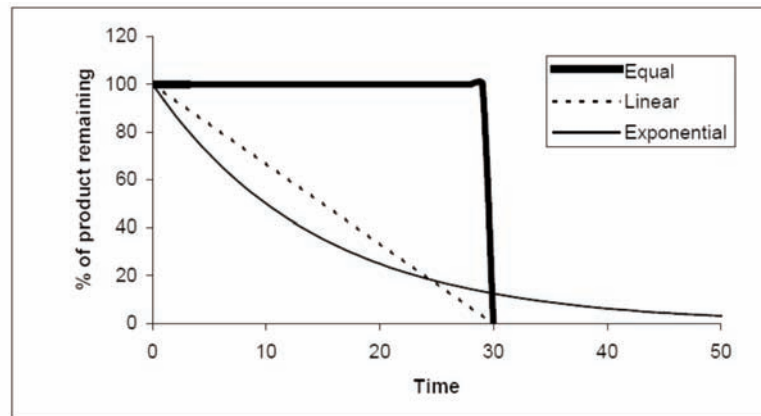
The different methods in modeling carbon release from HWPs clearly show how assumptions concerning product lifespan can significantly alter estimates.

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retirement most likely follows this decay function more closely as HWP retirement accelerates before reaching median life and finally slows down after the median life (Skog and Nicholson, 2000). It must be noted that these decay functions do not effectively model conditions in landfills or bioenergy facilities. Thus, they should only be used to model the rate of HWP retirement from use, which could then be incorporated into a larger model including a decay function that more accurately portrays carbon emissions from HWP once they are no longer in use.

**Figure 1** A graphical representation of how carbon release is modeled using different methods of incorporating HWP product life into stock calculations



Source: Pingoud, K., Soimakallio, S., Perala, A.L., Pussinen, A., 2003. Greenhouse gas impacts of harvested wood products: Evaluation and development of methods. VTT Technical Research Center of Finland, Research Notes 2189. Reprinted with permission.

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Industrial uses of HWPs such as pallets may provide better data on product life in the future as companies begin to label them with bar codes containing a pallet's age.

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Determining accurate HWP lifespan values in order to create models that simulate real life conditions is difficult given a lack of data from disposal facilities. There is room for vast improvement in reporting methods. Industrial uses of HWPs such as pallets may provide better data on product life in the future as companies begin to label them with bar codes containing a pallet's age. Carbon markets may also encourage companies to keep better data on product life as they may in the future be able to sell temporary carbon credits based on their HWP stock. Inclusion of HWPs in climate mitigation policy will require increased reporting which will lead to better data, allowing for more accurate product lifespans (Kuchli, 2008).

## REVIEW OF HWP CARBON STOCK STUDIES

Because HWPs do not last into perpetuity, the estimates of carbon sequestered in HWPs do not give an accurate picture of the rate at which carbon stocks of HWPs are increasing on an annual basis. Studies have shown that net annual increases in stocks of HWPs are estimated in a wide range from 26 to 139 Tg C/yr<sup>-1</sup> (IPCC, 2001; Matthews et al, 1996). This clearly demonstrates the importance of product life assumptions in calculating the yearly increase in the HWPs carbon stock. A common trend in many

studies is to show the HWP's carbon stock increasing at a sizeable annual rate. This increase could be attributed to greater demand for products due to population growth and increasing standards of living worldwide (Miner, 2008). Exclusion of end-of-life pathways creates high levels of uncertainty as different pathways can greatly alter product life and carbon emissions, which in turn will greatly impact estimates of annual increases of the HWP's carbon stock. Therefore it is imperative to incorporate end-of-life pathways into models despite arguments that data on end-of-use is unreliable.

**END-OF-USE PATHWAYS FOR HWPs**

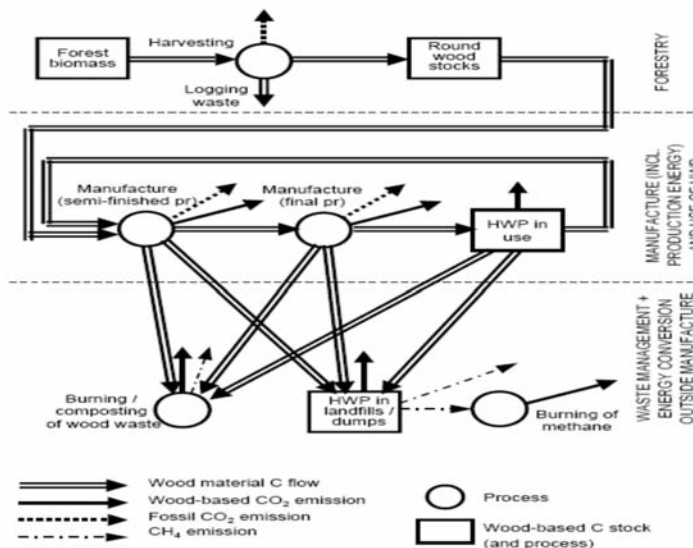
Harvested wood products can take several different pathways when they are discarded (CEPI, 2007). Recent research has expanded the system boundaries of analysis to account for the different end-of-use pathways which can postpone carbon release of HWPs, store carbon indefinitely, displace fossil fuels, or even produce emissions at a significant level. HWPs can be recycled, burned (with or without energy recovery), or disposed of in a dump or landfill (Figure 2). Each of these pathways has different implications for carbon emissions. Calculations that do not account for these pathways are not accurately capturing the carbon effects. This is especially true in regards to the production of CH<sub>4</sub> resulting from the land filling of HWPs. Research that includes end-of-use pathways has shown that from 2000-2005 the global HWP stock had an average net increase of 147 Tg C/yr, which is equivalent to 540 Tg CO<sub>2</sub>/yr (Miner, 2008). These findings are at the higher end of the range compared to earlier studies due to the study's assumptions on landfills.

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It is imperative to incorporate end-of-life pathways into models despite arguments that data on end-of-use is unreliable.

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Figure 2 Schematic representation of a lifecycle of HWP



Source: Pingoud, K., Soimakallio, S., Perala, A.L., Pussinen, A., 2003. Greenhouse gas impacts of harvested wood products: Evaluation and development of methods. Espoo 2003. VTT Technical Research Center of Finland, Research Notes 2189, 120 p. + app. 16 p. Reprinted with permission.

### Burning HWPs

HWPs have the potential to be burned as a fuel. Short-term wood products follow this pathway more often than long-term products. Skog and Nicholson (2000) estimate that in 1993 in the United States, over 24% of paper and paperboard waste (after recycling) was burned. Although burning discarded wood or paper for energy is a carbon emitting activity, it may result in lower net emissions if it has displaced more carbon-intensive fuel types (substitution effect). Using discarded HWPs for energy also reduces the amount that is put in landfills thus reducing the production of potent CH<sub>4</sub> gas.<sup>2</sup> In order to evaluate whether burning HWPs for energy is superior to burning an alternative energy, a comparison of the two fuel chains must use a consistent methodology and a consistent definition of system boundaries.

<sup>2</sup> CH<sub>4</sub> has a carbon multiplier of 24 per molecule, compared to CO<sub>2</sub>.

### Recycling

Recycling programs prolong the lifespan of carbon in HWPs, which keeps carbon stored in the product chain and extends carbon sequestration benefits. Recycling processes typically transform HWPs into products of lower wood content. This process can be repeated until the HWP is used to create bioenergy. This is known as a cascade effect (Kuchli, 2008). A high rate of HWP recycling can reduce the overall rate of landfilling. This then reduces the amount of CH<sub>4</sub> produced by HWPs in landfills (CEPI, 2007). This is particularly true for paper products, as these materials produce higher levels of CH<sub>4</sub> than solid wood products (Skog et al., 2004).

As HWPs cascade into products of lower wood densities, however, their viability to be recycled is reduced. Once paper has reached a very low grade, such as tissue, it can no longer be recycled. Not surprisingly, as a result of the cascading effect and the downcycling of HWPs, low-grade paper products constitute a third of municipal solid waste (MSW) in landfills (Pingoud et al., 2003; EPA, 2008).

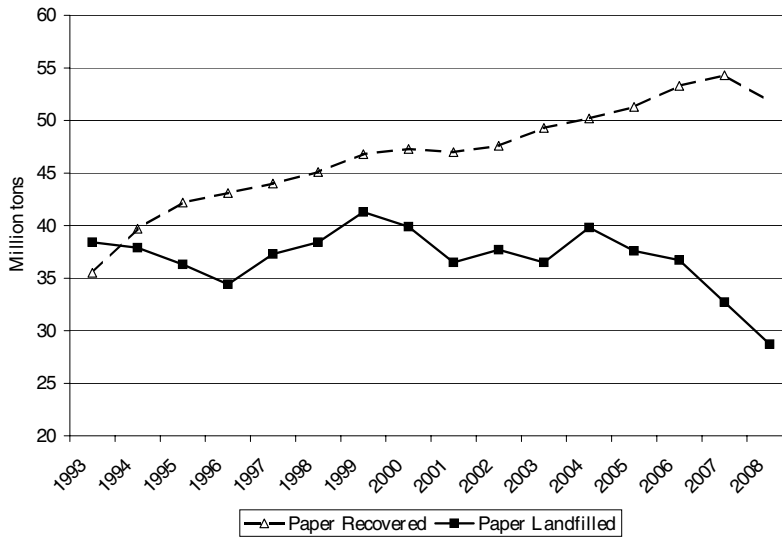
The type of HWP plays a major role in whether or not it will be recycled. At the moment, recycling is only seen as a viable option for paper products. The EPA reported that in 2007, 83 million tons (U.S.) of paper and paperboard were generated, of which 45 million tons (U.S.) (or 54%) were recovered through recycling (EPA, 2008). In contrast, the recycling rate for HWPs used in construction is significantly lower. In 2007, the United States recycled only 1.3 million tons of durable wood products from the nearly 14 million tons generated (9%) (EPA, 2008). This huge disparity in recycling rates is due to the nature of the products themselves. Newspaper is easily sorted and collected, while wood from construction demolition is very difficult to separate and re-use. Notably, data from the National Council for Air and Stream Improvement (NCASI) shows that while paper recovery is rising rapidly, the amount of paper products in landfills has decreased only nominally (Figure 3) (Miner, 2008). This suggests that recycling of HWPs may play a greater role in postponing landfilling and subsequent carbon emissions rather than simply reducing the amount of HWPs landfilled. Still, reductions in the amount of HWPs landfilled are expected to occur over time as HWP recycling processes modernize and become fueled by residue losses from the recycling process.

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Recycling programs prolong the lifespan of carbon in HWPs, which keeps carbon stored in the product chain and extends carbon sequestration benefits.

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Figure 3 Paper recovery vs. landfilling in the U.S. 1993 - 2007



Source: Created with data from the American Forest & Paper Association, [http://www.paperrecycles.org/stat\\_pages/recovery\\_vs\\_landfill.html](http://www.paperrecycles.org/stat_pages/recovery_vs_landfill.html).

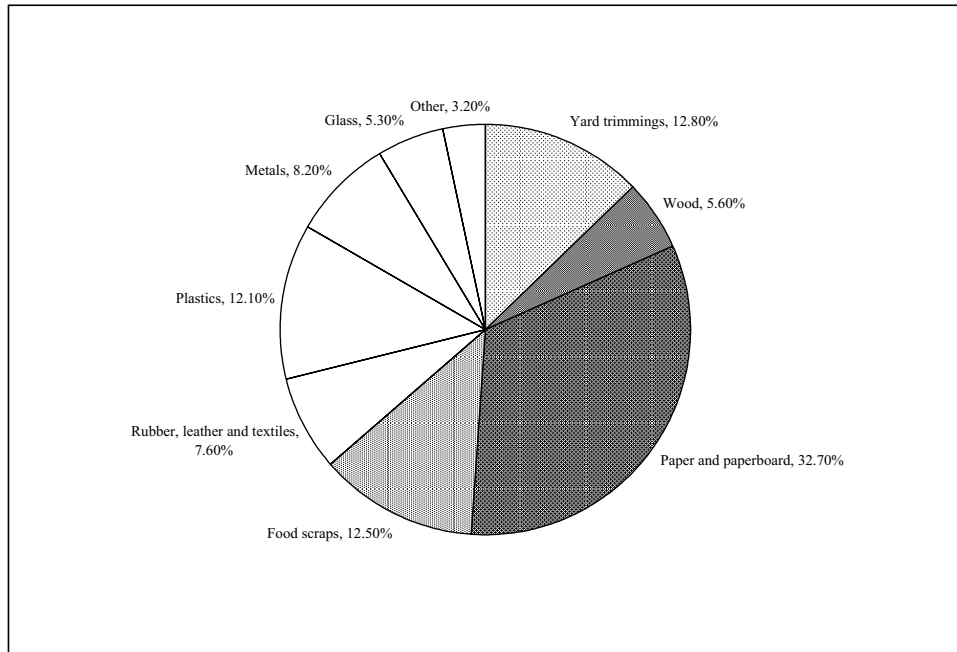
### Landfills

Landfills have been criticized for their negative environmental impacts since the beginning of the environmental movement. Today, however, there are those in the scientific community who suggest that landfills could potentially act as a carbon *sink* for HWP's due to the fact that HWP decomposition is shown to be very slow under anaerobic conditions in landfills (Green et al., 2006). Studies have shown that most wood products, when disposed of in a modern landfill, will stay there indefinitely with almost no decay (Bogner et al, 1993; Ximenes et al., 2008). This finding can have significant implications for calculating the stock of carbon in HWP's because it is estimated that biomass materials, such as paper, food, and wood, constitute about 63% of the municipal solid waste (MSW) (Figure 4) (EPA, 2008). The high proportion of HWP's in landfills further supports the case to expand the boundaries of analysis to include HWP end-of-use pathways. If  $\text{CH}_4$  is captured and used for energy, carbon emission reductions can occur as carbon remains locked in HWP's at the same time that energy generated from landfill gases can displace fossil fuel emissions from traditional energy use. Despite the attractiveness of using landfill gases for fuel, recent estimates indicate that only around 5 Tg C is captured worldwide, versus 15-20 Tg C of annual emissions from landfills (Spokas et al., 2006). The large discrepancy between landfill gas (LFG) production and capture is best understood by analyzing how landfills work as well as current disposal practices. This may also help to forecast the likely impact from policies under debate to encourage increased landfilling of HWP's.

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The type of HWP plays a major role in whether or not it will be recycled. At the moment, recycling is only seen as a viable option for paper products.

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**Figure 4** Materials generated in Municipal Solid Waste in the United States, 2007.

Source: Data from EPA, 2008. Municipal solid waste in the United States: 2007 Facts and Figures. U.S. Environmental Protection Agency, Office of Solid Waste, EPA530-R-08-010.

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There are those in the scientific community who suggest that landfills could potentially act as a carbon sink for HWP due to the fact that HWP decomposition is shown to be very slow under anaerobic conditions in landfills.

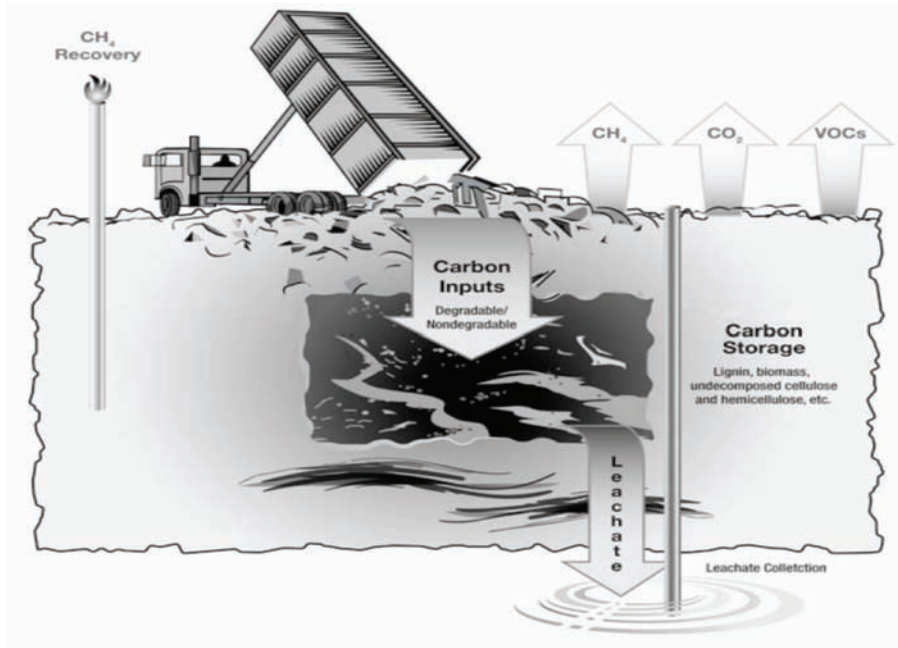
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### Landfill science

In a landfill, solid waste is buried. While this allows some biodegradable fractions of the waste to decompose via a complex series of microbial and abiotic reactions, the anaerobic conditions prevent a significant amount of decomposition.  $\text{CH}_4$  is formed by methanogenic microorganisms under anoxic conditions, either through the direct cleavage of acetate into  $\text{CH}_4$  and carbon dioxide or the reduction of  $\text{CO}_2$  with hydrogen (Barlaz et al, 1990, Tong et al, 1990; Spokas et al., 2006).

Since new layers of waste cannot be instantly covered, the waste is exposed to oxygen which allows white-rot fungus to occur and decay wood. This type of decay, however, is limited due to the fact that the available oxygen is rapidly consumed by the fungus, leaving only anaerobic bacteria. While anaerobic bacteria can break down hemicellulose and cellulose, these organisms cannot reach these materials if they are enclosed in lignin (Skog and Nicholson, 2000). As a result, solid wood placed in landfills experiences low rates of decay. In newsprint, however, lignin content is only 20-27%, which results in a greater risk of decay than solid wood products, despite anaerobic conditions. Still, both wood and paper products experience low decay rates; in general, less than 50% of the carbon in these products is ultimately converted to  $\text{CO}_2$  or  $\text{CH}_4$  (Table 2) (Skog and Nicholson, 2000).

Figure 5 Landfill carbon mass balance



Source: EPA, 2006. Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, 3rd edition. United States Environmental Protection Agency.

Table 2 Estimated maximum proportions of wood and paper that are converted to CO<sub>2</sub> or CH<sub>4</sub> in landfills.

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

Source: Skog, K., Nicholson, G.A., 2000. Carbon Sequestration in Wood and Paper Products. In: Joyce, L.A., Birdsey, R., technical editors. 2000. The impact of climate change on America's forests: a technical document supporting the 2000 USDA Forest Service RPA Assessment. Gen. Tech. Rep. RMRS-GTR-59. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 79-88

Emissions created from anaerobic conditions are referred to generally as “landfill gas” (LFG) and encompass multiple gases, predominantly CO<sub>2</sub> and CH<sub>4</sub> (Figure 5). According to Skog and Nicholson (2000), the proportion of carbon that is emitted as CO<sub>2</sub> and CH<sub>4</sub> (Figure 5) in the gaseous product of MSW in landfills is skewed towards CH<sub>4</sub> at a rate of 1.5:1. Other studies show that the proportional difference between the two is not as great and that 1:1 should be used for commercial purposes (Johannessen, 1999; Themelis and Ulloa, 2007).

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It is also important to note that emissions of various greenhouse gases occur on different temporal scales. On the one hand, CO<sub>2</sub> is released quickly as decomposition occurs while oxygen is still present in the system. Studies estimate that half of the total CO<sub>2</sub> is emitted in the first 3 years while the rest is emitted continually over time (Skog and Nicholson, 2000). Methane, on the other hand, is released very slowly over time once all the oxygen is depleted, with half the total CH<sub>4</sub> emitted in approximately 20 years (Micales and Skog, 1997). Moreover, Skog and Nicholson claim that 10% of the CH<sub>4</sub> is converted to CO<sub>2</sub> by micro-organisms as it moves out of the landfill, which makes the landfill cover a de facto converter. According to Johansson, the conversion capacity for a landfill top cover varies depending on soil texture, moisture content, and the amount of organic matter available in the soil. Covers with porous soils and organic matter have achieved complete oxidation of methane (Johannessen, 1999).

While LFG production poses a problem in terms of carbon emissions, high LFG production levels are desirable for LFG capture system operators, particularly since such systems are capital intensive and often financed by energy sales. Although theoretically, one ton of biodegradable carbon can produce 1,800 m<sup>3</sup> of LFG, in practice, this number is much lower in most cases because of uneven and incomplete biodegradation. As a result, 200 m<sup>3</sup> is generally accepted as the maximum volume of LFG produced from one ton of land filled MSW (Johannessen, 1999). Several factors influence the rate of capture to total volume of LFG generated. These include LFG losses to the atmosphere through the surface or through lateral gas migration; pre-closure loss due to decomposition of organic material under aerobic conditions; aerobic decomposition of the near-surface layer (e.g., air intrusion due to gas extraction); and washout of organic carbon via leachate (Johannessen, 1999). All of these can reduce the potential LFG capture rate, and often tip the balance of whether landfills reduce emissions from carbon storage or serve as large sources of emissions.

There are more than 350 landfills in the United States with gas recovery plants, and more than 1,100 worldwide (Spokas et al., 2006). These landfills are very diverse with respect to the amounts of material placed in the landfill, the type of material, degradation rate, and LFG capture system. Moreover, within individual landfills, decomposition rates can vary even in adjacent areas of a landfill (Micales and Skog, 1997). This variation makes it difficult to assign an average capture rate to all landfills (CEPI, 2007). As one example, the EPA's Waste Reduction Model (WARM) uses a default value of 75% LFG capture rate. Compared to other reports, this figure is higher than average and likely varies greatly from region to region within the United States (Themelis and Ulloa, 2007). Other studies are more conservative and claim that normal recovery rates are thought to range from 40% to 50% by volume (Johannessen, 1999). In this case, even landfills with advanced cover systems are thought to recover just slightly over 60% of the LFG produced. However, a more recent study in France found LFG recovery rates ranged from 41% to 94% of the theoretical CH<sub>4</sub> production and were highly dependent on the engineered cover design (ADEME, 2008). It further suggested that average LFG recovery rates could exceed 90% by excluding the poorest performing cover design from the study.

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LFG generation and capture rates vary across a temporal scale. This has led the French environment agency (ADEME) to create different default values to account for landfill design and stage of operation with values ranging from 35% to 90% recovery (Spokas et al., 2006). The literature on this subject clearly shows that there is a high level of uncertainty when it comes to calculating emissions from landfills. However, industry experts believe that methane emissions from wood products in landfills will become a smaller part of the total carbon footprint from HWP as technology improves and more LFG is captured (Miner, 2008).

In 2007, 3.7 billion m<sup>3</sup> of methane was captured from landfills in the United States, of which 70% was used to generate thermal or electrical energy (Themelis and Ulloa, 2007). The rest of the methane was flared since it was thought to have no economic value. Flaring of LFG and using it in energy production reduces the methane content to carbon dioxide and water (Johannessen, 1999). Despite the fact that flaring reduces the potency of the methane, it still produces high levels of CO<sub>2</sub> emissions. It must also be noted that there are nearly 1,400 landfills in the United States (EPA, 2008) that do not capture and flare any biogas. It is likely that HWPs in these sites are generating high levels of CH<sub>4</sub> emissions.

Including end-of-use conditions in HWP carbon stock models is critical due to the large potential emissions from landfills. Carbon released during end-of-use processes does not follow the simple decay functions most often used to model HWP retirement and discard. As shown earlier, landfills may have varying conditions which will have a large impact on HWP carbon stocks. How these landfills are incorporated into HWP carbon stock accounting is key. In the United States, for example, only 20% of 1,754 landfills are currently capturing LFG (EPA, 2008). This figure raises serious doubts on the default LFG capture rate of 75% used by the EPA in the WARM model. Unfortunately, unrealistic default LFG capture rates have the potential to greatly miscalculate the role of HWPs not only on a country basis but globally. Policies that promote landfilling of HWPs must therefore be aligned with policies that require high percentage LFG capture rates to ensure net emission reductions.

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A recent study in France found LFG recovery rates ranged from 41% to 94% of the theoretical CH<sub>4</sub> production and were highly dependent on the engineered cover design.

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## CONCLUSIONS AND MANAGEMENT AND POLICY IMPLICATIONS

The production and use of HWPs may postpone carbon emissions as carbon is stored in HWPs for a period after the initial harvest of roundwood. If the production of HWP exceeds the rate of retirement, then the amount of carbon bound in the HWP stock increases. This is supported by the literature.

### Management and policy implications

- With global climate mitigation policy calling for the protection and enhancement of carbon sinks, policies to enhance the HWP carbon stock should be considered.
- Current methods for estimating carbon in HWPs are highly variable. A lack of data on product use makes it difficult to model HWP stocks; even



assumptions on average wood density can significantly alter estimates of the conversion of HWP mass into carbon.

- The way carbon release of HWPs is modeled also has significant impacts on estimates. Recycling should be promoted heavily in policy intended to enhance the HWP carbon stock since recycling postpones carbon emissions of even short-lived HWPs. Recycling also fits very well in the “cascaded use of HWPs” concept where HWPs are transformed multiple times within a tight recycling chain and finally converted into bioenergy.
- Landfills could potentially be an attractive final destination for discarded HWPs since HWPs have shown to have very low rates of decay in landfills. The production of LFG also fits into the “cascaded use of HWPs” framework because it can be converted into energy, displacing fossil fuels and further reducing global emissions.
- It must be demonstrated that unintended consequences are not triggered by policies intended to enhance the HWP carbon stock. Policies that promote landfilling HWPs must recognize that landfills create high levels of methane, and if capture systems for energy are not in place, then the potential of landfills acting as a carbon sink becomes very unlikely. Therefore, landfill gas (LFG) capture systems must be required if this end-of-use pathway is to be promoted as a way to reduce carbon emissions.

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## Part III: Socio-economic and Policy Considerations for Carbon Management in Forests

### SECTION SUMMARY

While the biophysical characteristics of forests covered in the earlier parts of this book define the boundaries within which forest management can occur, actual management practices are driven by economic, policy and other cultural values. The purpose of the following chapters is to explore the economic and policy drivers that affect the opportunities for managing forests with carbon in mind. In the first three chapters, the economic pressures and incentives facing land managers are described: first, in tropical developing countries experiencing rapid rates of deforestation as land is converted to more remunerative agricultural uses; second, in tropical countries retaining large areas of relatively intact forests as a result of physical or market isolation; and third in the U.S., where the economics of developing land for buildings far outweighs the incentives for maintaining land as farms and forests. Finding ways to use policy to help overcome these incentives for land managers to convert forests to more lucrative uses of the land is the focus of the last two papers. The factors to be considered when deciding between use of the carbon markets (through offset projects) or direct public funding of forest conservation are described at both the global level as part of the REDD+ negotiations and at the federal level in the U.S. building on the experience in the voluntary carbon markets. While increasing numbers of people agree that forests and other land use issues have to be a significant part of the global response to climate change, the ways in which this goal will be achieved are still open to considerable debate.

Contributors toward organizing and editing this section were: *Bradford Gentry, Deborah Spalding, Mary L. Tyrrell, and Lauren Goers.*

## Chapter 14

# Economic Drivers of Tropical Deforestation for Agriculture

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### EXECUTIVE SUMMARY

Land use change from deforestation in the tropics is increasingly recognized as a major source of greenhouse gas emissions. In order to develop policies that address this significant portion of emissions that contribute to global climate change, it is essential to understand what drives deforestation in the tropics. This chapter examines the socioeconomic, institutional and economic drivers of tropical deforestation for agriculture in order to gain a better understanding of how incentives to sequester carbon in forests may impact deforestation rates. While the circumstances that drive deforestation are locally based and depend upon a variety of factors that include social, political and geographical considerations, there are some general lessons that can be learned from our review of the literature. For example, at a local scale, population pressure and poverty can be shown to lead to deforestation, but these explanations are limited in their ability to describe the *scale* of deforestation that many tropical countries have experienced in recent years. Development efforts such as road-building into forested areas are significantly correlated with deforestation throughout the tropical region. Institutional factors such as land tenure laws that incentivize forest clearing or macroeconomic policies that provide agricultural subsidies have also been shown to influence deforestation rates in countries such as Brazil. However, in most regions, the factors leading to deforestation are complex and interrelated. The complexity of these drivers has significant implications for global climate negotiations where the international community seeks to negotiate a mechanism to reduce emissions from deforestation and forest degradation (REDD). In order to build a successful REDD mechanism, it will be essential to develop avoided deforestation strategies that incentivize countries

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to address these underlying factors, including the necessary economic and institutional reforms.

**Keywords:** deforestation, reduced emissions from deforestation and degradation (REDD), agriculture, carbon sequestration, land use change, agroforestry

## INTRODUCTION

Fossil fuel combustion is frequently cited as the main culprit of human-induced climate change (Barker et al., 2007; Betts et al., 2008). Although the extraction and use of fossil fuel are indeed large contributors to greenhouse gas emissions, the impacts of land cover change and deforestation, particularly in the tropics, also account for a significant percentage of annual greenhouse gas emissions, estimated at approximately 17.4% by the IPCC (Barker et al., 2007).

While fossil fuel emissions come primarily from developed nations with high levels of industrialization, consumption, and vehicle use, emissions from land use change and forestry largely stem from cutting down tropical forests for agriculture in developing countries such as Brazil and Indonesia (FAO, 2005). Altering forest and land management practices in these regions is often seen as an important component of efforts to reduce global greenhouse gas emissions, and one that may be quicker and less expensive to implement than restructuring the economies and infrastructure of developed countries. It is also thought to have other benefits beyond climate change amelioration, including providing funding for capacity-building and technology transfer to developing countries to help implement changes in forest management practices.

This chapter reviews the current research on the drivers of deforestation in the tropics with a focus on land clearing for agriculture. It considers the impact of socioeconomic, institutional and economic factors on rates of deforestation for agriculture, particularly in key countries with large emissions from forests such as Brazil and Indonesia. It concludes by outlining several issues that policymakers and land managers must consider when developing incentives to prevent greenhouse gas emissions from forest conversion and degradation.

Currently, emissions from land-use change activities are estimated to produce 17.4% of greenhouse gas emissions (Figure 1), largely from deforestation. For the purposes of this chapter, deforestation is defined as the conversion of forest to another land cover when tree canopy falls below a certain established minimum threshold (Lepers et al., 2005). The Food and Agriculture Organization uses a tree canopy cover of 10% to classify areas as forested (FAO, 2005).

Land clearing of tropical forests for agriculture is a significant portion of the total greenhouse gas emissions from forestry and a primary driver of tropical deforestation (Angelsen, 1995). The significant amount of land clearing is attributable in part to the fact that nearly 700 million people live near tropical forests and depend on forest land or resources for food, fuel and a source of income (Chomitz et al.,

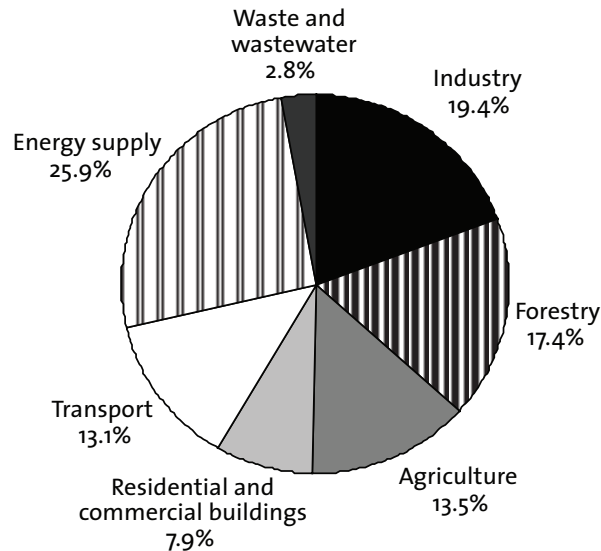
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2007). The conversion of forest land into agricultural systems, including swidden agriculture practiced by indigenous groups, subsistence farming by smallholders, commercial use as industrial plantations, and pasture land all contribute to the rapid loss of tropical forests worldwide (Barbier and Burgess, 2001).

**Figure 1 Greenhouse gas emissions by sector**



Source: Betts, R., Gornall, J., Hughes, J., Kaye, N., McNeill, D., Wiltshire, A. 2008. Forest and Emissions: Contribution to the Eliasch review. <http://www.occ.gov.uk/activities/eliasch.htm> Accessed 25 February 2009. Reprinted with permission.

### Deforestation trends

To understand the complex relationship between deforestation and agriculture, and to better implement carbon policies aimed at reduced deforestation, it is important to identify where and at what rate deforestation is occurring. During the 1980s, FAO estimated that nearly 15.4 million hectares of tropical forests were cleared each year (Angelsen and Kaimowitz, 1999). Subsequent studies have shown a slight decrease in overall forest loss in the 1990s, but changing definitions of forest could account for some of that loss (Angelsen and Kaimowitz, 1999).

Worldwide figures showing the scale of forest loss are important tools for understanding the magnitude of the problem. However, examining regional variations in forest loss is important for studying the underlying drivers of deforestation in different regions of the world. The world's three major tropical regions differ in amount and rate of forest loss (Table 1). According to Achard et al. (2002), Southeast Asia has the highest rate of tropical forest conversion for the period spanning 1990-1997. Although deforestation rates in Africa and Latin America are lower within that time frame, the total area of forest converted is similar in Latin America and Southeast Asia.

Degraded forest lands, defined as forests where changes have negatively altered the structure or function of the site (including the capacity to sequester carbon), show a similar trend (FAO 2005) – the change in area of degraded forest is highest in Southeast Asia, followed by Latin America and Africa.

**Table 1 Humid tropical forest and annual changes 1990-1997 (millions of hectares)**

	Latin America	Africa	Southeast Asia	Global
Total study area	1155	337	446	1937
Forest cover in 1990	669 ± 57	198 ± 13	283 ± 31	1150 ± 54
Forest cover in 1997	653 ± 56	193 ± 13	270 ± 30	1116 ± 53
Annual deforested area	2.5 ± 1.4	0.85 ± 0.30	2.5 ± 0.8	5.8 ± 1.4
Rate	0.38%	0.43%	0.91%	0.52%
Annual regrowth area	0.28 ± 0.22	0.14 ± 0.11	0.53 ± 0.25	1.0 ± 0.32
Rate	0.04%	0.07%	0.19%	0.08%
Annual net cover change	-2.2 ± 1.2	-0.71 ± 0.31	-2.0 ± 0.8	-4.9 ± 1.3
Rate	0.33%	0.36%	0.71%	0.43%
Annual degraded area	0.83 ± 0.67	0.39 ± 0.19	1.1 ± 0.44	2.3 ± 0.71
Rate	0.13%	0.21%	0.42%	0.20%

Source: Achard, F., Eva, H.D., Stibig, H.J., Mayaux, P., Gallego, J., Richards, T., Malingreau, J.P., 2002. Determination of deforestation rates of the world's humid tropical forests. *Science* 297, 999-1002. Reprinted with permission from AAAS.

Tropical deforestation for agricultural purposes has significant implications for local, regional and global climate trends. As noted above, forestry (mostly conversion of forests to other land uses) contributes 17.4% of global greenhouse gas emissions. Coupled with emissions from agriculture at 13.5%, total land use activities generate nearly one third of global emissions. Since tropical forests account for approximately 37% of the world's forested area, they are also a critical carbon sink (Betts et al., 2008). Continued deforestation of tropical forested ecosystems has the potential to release vast amounts of stored carbon, which would have significant ramifications for global climate.

### Conversion to different types of agricultural land uses

Landowners convert forested land to a variety of different agricultural uses. Often their decision is based on a combination of site characteristics and economic, political, and social drivers. Sixty-nine percent, or 3,488 million hectares, of the 5,023 million hectares designated worldwide as agricultural land are used for pasture or forage crops (Smith et al., 2007; Lambin et al., 2003). Lands with marginal productivity typically do not generate significant return on the investment of capital and labor for growing crops, and are therefore converted to less intensive agriculture, including pasture (Lambin et al., 2003). In contrast, intensive agriculture is often placed on higher quality, more productive lands. While intensive agriculture supports increased food production, it often also has higher input requirements per unit of



area, relying upon mechanization, fertilizers and agrochemicals. Agroforestry systems are mixed systems that can combine trees, shrubs, crops, grasses, and animals and may have high carbon sequestration potential compared to other productive land use options (Ilany and Lawson, 2009). Fallow lands are agricultural lands that have been idle for one or more growing seasons.

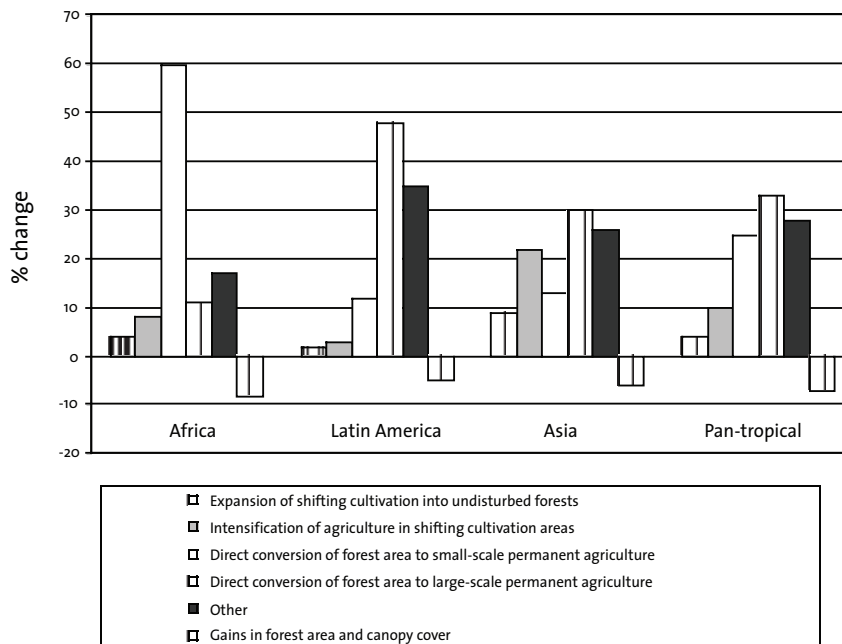
While land conversion itself is a significant source of carbon emissions, carbon may be sequestered once agricultural systems are implemented. Carbon sequestration rates, however, will differ depending on the type of agriculture and the productivity of the site. In a study comparing the potential of different land use systems to sequester carbon in eastern Panama, managed forests were found to store an average of 335 Mg C per ha, traditional agroforestry systems stored an average of 145 Mg C per ha, and pastures stored an average of 46 Mg C per ha (Kirby and Potvin, 2007). Another study in Saskatchewan, Canada, compared the median ecosystem carbon density for forests, pastures, and cultivated fields (Fitzsimmons et al., 2004). Forest ecosystems contained a median of 158 Mg C per ha, while pastures contained 63 Mg C per ha and cultivated fields contained 81 Mg C per ha (Fitzsimmons et al., 2004). The level of carbon sequestration within a forestry or agricultural system varies between sites in relation to different biophysical characteristics and climatic variations, as well as the different land use and management techniques.

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**Figure 2** Percentage of total area change by land use type, 1980–2000



Source: Derived from Martin, 2008

There is a clear regional variability in the types of agricultural management implemented on deforested land. While Latin America is dominated by deforestation

for livestock and pasture lands, forest conversion in Africa is characterized by small farm croplands (Lambin et al., 2003). In Asia, forest loss is attributable to both widespread logging and the establishment of permanent agricultural crops (Lambin et al., 2003; Kummer and Turner, 1994).

Shifts in land use between forest and agricultural systems are often dynamic. After initial deforestation, land is typically used by farmers in the Brazilian Amazon for annual crops for an average of two years, then either shifts to pasture or perennial crops, or is left fallow (Vosti et al., 2001). It is estimated that the conversion of Brazilian tropical rainforest to arable land releases 703-767 Mg CO<sub>2</sub> equivalent per hectare (Reijnders and Huijbregts, 2008). The remote sensing data in Figure 2 compare changes in land use between 1980 and 2000 in Africa, Asia, and Latin America (Martin, 2008). Looking at the differences in land use and forest conversion to agriculture in these regions, it appears that a variety of interrelated factors drive regional differences. These are explored in the following section.

### DRIVERS OF TROPICAL DEFORESTATION

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A review of the literature indicates three major categories of deforestation drivers in the tropics: socioeconomic, institutional, and economic factors

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Where there is general acceptance that agricultural conversion accounts for a significant amount of tropical deforestation, the mechanisms that drive conversion of forests for agriculture are less clear. Academic debate has ranged from simple, single driver hypotheses such as population growth or poverty as main causes of land use conversion to more complex models that list combinations of market-based explanations and other socio-political factors (Geist and Lambin, 2002). Econometric models and empirical studies are often used in an attempt to explain the combination of factors that drive deforestation in an effort to design better policies that will slow forest loss while addressing the underlying causes of encroachment into forest areas. A review of the literature indicates three major categories of deforestation drivers in the tropics: socioeconomic, institutional, and economic factors (Figure 3).

**Figure 3 Drivers of deforestation for agriculture**

Socioeconomic Factors	Institutional Factors	Economic Factors
<ul style="list-style-type: none"> <li>• Population Growth</li> <li>• Urbanization</li> <li>• Poverty and Economic Inequalities</li> <li>• Transportation Infrastructure</li> <li>• Agricultural Technology</li> </ul>	<ul style="list-style-type: none"> <li>• Property Rights and Land Tenure</li> <li>• Governance Regulatory Enforcement, Corruption, and Political Stability</li> </ul>	<ul style="list-style-type: none"> <li>• International Trade and Economic Integration</li> <li>• National Economic Policy</li> <li>• Household and Local Economies</li> <li>• Household-Level Decision Making</li> </ul>

## Socioeconomic factors

### *Population growth*

Population growth is frequently cited as a major driver of deforestation for agriculture in the developing world (Lambin et al., 2001; Allen and Barnes, 1985). However, the argument that population growth adequately explains deforestation rates is not as robust as previously thought. Researchers frequently attribute tropical deforestation to increasing populations of shifting agriculturalists, despite the fact that recent FAO data estimate that shifting cultivators account for only 5% of pan-tropical forest conversion (Chomitz et al., 2007). Some models find a correlation between population growth and clearing of forest land at the national level. However, subsequent analyses reveal that populations moving into forested areas and subsequently clearing land is a function of a host of other factors that include access to infrastructure, high quality soils, off-farm employment opportunities and distance to markets (Angelsen and Kaimowitz, 1999). The relationship between population figures and the pressure populations exert on forest resources is difficult to parse without examining the underlying causes of population growth.

More recent work focuses less on the impacts of overall population growth and instead seeks to characterize deforestation trends as they relate to different population types. For example, Jorgenson and Burns (2007) focus on patterns in urban and rural population growth, migration patterns, and economic development to draw contrasts between the location of population growth and the impacts on forest cover (Jorgenson and Burns, 2007). Their results indicate that while rural population growth does drive deforestation, urban population increases actually have a slowing effect on forest conversion for agriculture as subsistence farmers migrate to urban centers for work. Other work on population and deforestation suggests that the location of population growth is significant; the first people entering a frontier area have much more impact on deforestation in an area than population growth or migration in an already populated area (Pfaff, 1996). These findings may be significant for forest policy, as they indicate the importance of spatial heterogeneity of population density in addressing deforestation rates and drivers.

### *Urbanization*

Urbanization and the movement of populations are also important to consider when examining relationships between population and deforestation. While urban areas tend to be more compact and require less land, changing urban diets and consumption patterns ultimately lead to a greater strain on rural natural resources. Additionally, land use change from urban areas frequently expands into nearby agricultural land, thus pushing agricultural pressures into forested areas (Lambin et al., 2003). Overall, the impacts of urbanization on land use change in forests need to be studied more closely on a local level. Urbanization trends lead to complex and non-linear feedback mechanisms that include rural encroachment, the migration of landless workers from urban centers back to rural areas, or abandonment of agricultural lands that leads to secondary growth (Jorgenson and Burns, 2007).

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Case studies of the impacts of population on forest cover reveal the complexities associated with determining what drives deforestation in an area. For example, population growth on the Indonesia island of Java led the Indonesian government and the World Bank to sponsor a transmigration program that transplanted Javanese urban dwellers to the more remote, largely forested islands of Kalimantan (Borneo) and Irian Jaya (West Papua). These government policies to reduce population density in urban areas had important implications for deforestation in Indonesia during the late 1970s and early 1980s (Fearnside, 1997). The lack of traditional agricultural knowledge on the part of the migrants and the influx of spontaneous migrants who were not part of the government-sponsored program led to increased deforestation for agricultural uses. Subsequent government-sanctioned migration programs in Indonesia have increased migration to outer islands as a means of subsidizing labor for timber plantations, primarily oil palm. The impacts of transmigration policy on forests in Indonesia are estimated to range from 2-5 hectares per family for the early programs that encouraged subsistence farming, to nearly 20 hectares per family for industrial plantation farming (Fearnside, 1997).

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### *Poverty*

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While urban areas tend to be more compact and require less land, changing urban diets and consumption patterns ultimately lead to a greater strain on rural natural resources.

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Like population, the poverty hypothesis has traditionally been cited by scholars as a key reason that deforestation for agriculture occurs in developing countries. Poorer farmers have more of an incentive to deforest in the short term rather than waiting for longer term potential profits from other land uses (Lambin et al., 2001; Angelsen and Kaimowitz, 1999). However, this view results in an oversimplification that attributes much of the deforestation for agriculture in tropical countries to poor smallholders, rather than to larger industrial plantations, government-sponsored concessions or other macro-scale land uses (Angelsen, 1995).

An alternate view of the poverty hypothesis contends that smallholders do clear some of the forest for subsistence, but they lack the capital, labor, and access to credit that is required to invest in large-scale forest clearing (Angelsen and Kaimowitz, 1999). This conclusion supports the finding of Chomitz et al. (2007) that conversion of forest to large-scale agriculture accounts for approximately 45% of land clearing in Asia and 30% in Latin America, whereas shifting cultivation of smallholders only accounts for approximately 5% of forest clearing. The situation in tropical Africa does differ somewhat, as over half of land use change is attributed to forest clearing for permanent, small-scale agricultural endeavors (Chomitz et al., 2007). While the reasons for these regional differences are complex, one contributing factor is the high global demand for the timber species found in the Asian and Latin American forests as compared to the African tropical forests.

A meta-analysis of research into poverty-related causes indicates that there are localized cases in which poverty is a driving force of deforestation. In these cases it is frequently driven by subsistence needs of smallholders and a desire for income maximization (Geist and Lambin, 2003). Yet, the impact of poverty on forest clearing is not straightforward. Similar to population factors, it is more likely that deforestation is driven by the combined social, economic and political factors that

contribute to poverty in poor countries. This conclusion is supported by research that shows that, in those cases where poverty has been a driving factor of deforestation, it is nearly always associated with other causes such as state incentives promoting land clearing, population growth and insecure land tenure (Geist and Lambin, 2003).

### *Economic inequalities*

Due to economic inequalities on a local and regional scale, access to economic opportunities, technology, and land differs across households and regions. During the 1970s, subsidized credit for machinery and chemical inputs for soybeans in Brazil was given primarily to large land owners (Kaimowitz and Smith, 2001). Not surprisingly, the high commodity price of soy and subsidized credit led to increases in land prices. Facing high land costs, expensive machinery, and chemical inputs for producing mechanized soy, small farmers could not compete, resulting in land consolidation by large operators (Kaimowitz and Smith, 2001). Estimates indicate that in Brazil, the expansion and mechanization of soybean production leads to the displacement of eleven farm workers for every worker employed (Altieri and Bravo, 2006). With a total of almost 3 million people displaced by soybean production in the Brazilian states of Parana and Rio Grande do Sul in the 1970s, many of these displaced individuals moved to the Amazon and subsequently cleared forest for agriculture (Altieri and Bravo, 2006). In this way, disproportionate access to economic opportunities, technology, and land at the expense of small landowners exacerbated income inequalities and further increased deforestation trends.

### *Transportation*

Regardless of the type of study or methodology used, roads are frequently shown to be highly correlated with an increase in deforestation, including roads for agricultural purposes (Angelsen and Kaimowitz, 1999). Increased infrastructure allows greater access to interior forests and to end markets for products. While there is a general consensus in the literature that increased access will lead to less forest, roads are both facilitators of deforestation activities as well as by-products of other economic activities that may already be causing deforestation (Lambin et al., 2003, Angelsen and Kaimowitz, 1999). In some cases, such as in central Africa, roads built for logging concessions typically lead to an influx of new residents who may clear the forest for agriculture (Burgess, 1993). While roads are considered a driver of deforestation in most tropical areas, there are some noteworthy exceptions. For example, areas with low population density or pressure from growth such as West Kalimantan, Indonesia, do not show a strong correlation between the presence of paved roads and deforestation pressure (Curran et al., 2004). Whether this is a short term observation or a trend that will continue longer term is unknown, however.

The role that roads have in the landscape varies by geography and other factors. In the case of West Kalimantan, the high value of dipterocarp timber species and the power of the timber industry in the region have a much stronger impact on deforestation than the presence of either roads or people. Additionally, roads can

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In those cases where poverty has been a driving factor of deforestation, it is nearly always associated with other causes such as state incentives promoting land clearing, population growth and insecure land tenure.

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promote connectivity between rural areas and nearby towns offering jobs that reduce the need for individuals to clear forestland for income (Chomitz et al., 2007). Thus, although roads are a primary driver of deforestation in most parts of the tropics, their local impact can be variable.

### *Technology*

Depending on the local economy, technologies that increase agricultural productivity have generally been associated with both forest loss and avoided deforestation. While several hypotheses have explored the causal links between technology and deforestation, two in particular stand out. The Borlaug hypothesis asserts that new higher-yielding technologies can increase agricultural production and profitability, thereby reducing deforestation pressures (Angelsen and Kaimowitz, 2001). Although this hypothesis might prove true for global food production, it has been shown that commodity prices have a greater impact on agricultural expansion than technological change at the local and regional levels, and particularly on forest frontiers. The economic development hypothesis holds that increased agricultural productivity due to technology will enhance overall economic development, thus decreasing poverty and the pressure on forests. (Angelsen and Kaimowitz, 2001).

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The mechanization of agricultural production can lead to land degradation due to soil erosion, compaction, and loss of fertility, thus increasing pressure on forests for agricultural land conversion.

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The impacts of agricultural technology on deforestation depend on a myriad of factors, including farmer characteristics, the scale of adoption, how the technology impacts labor and migration, and the profitability of agriculture on the forest frontier (Lambin et al., 2003; Angelsen and Kaimowitz, 2001). Technologies that allow farmers to save capital and to create jobs, while also driving increased productivity, will be most successful at diminishing pressures on forests. However, the mechanization of agricultural production can lead to land degradation due to soil erosion, compaction, and loss of fertility, thus increasing pressure on forests for agricultural land conversion. The industrialization of agriculture can also lead to land consolidation and loss of rural employment, leading to the displacement of small farmers and farm workers to marginal lands or the forest frontier (Lambin et al., 2003). In the Brazilian Amazon, mechanized agriculture increased by more than 3.6 million hectares between 2001 and 2004, mainly for soybean plantations. As a result, cattle ranchers have been displaced and are increasingly pressuring the forest frontier (Azevedo-Ramos, 2007). The complexity of factors affecting technological innovation and adoption, as well as the diversity of consequences resulting from such innovation, can lead to either an increase or a decrease in the rate of forest loss.

## **Institutional factors**

### *Land tenure*

Property and land tenure rights are another important driver of deforestation for agriculture. Many countries with high rates of deforestation and agricultural production are in countries that are still developing economically and may have weak institutional governance and forest law enforcement.

There are often high rates of deforestation in countries where clearing the land is the primary mechanism for claiming property rights (Angelsen and Kaimowitz,

1999). For example, land tenure laws in the Brazilian Amazon incentivize deforestation by granting title to settlers who “improve” the land by clearing forests (Mendelsohn, 1994). The impetus to clear the land as a means of securing title may also interact with road building and technology transfer since it provides an incentive for smallholders to anticipate road development or economic growth by clearing an area before anyone else.

Studies that relate land tenure security to deforestation have found that, in some instances, even secure tenure is not enough to stop forest clearing completely (Angelsen and Kaimowitz, 1999). In order for the landowner to see forest preservation as a viable management option, the financial benefits of keeping the forest intact must still outweigh the net present value of clearing the forest for agricultural production. However, longer term, the relationship between land tenure and forest clearing will ultimately depend on factors such as enforcement and governance. Regional level studies in Latin America, for example, have shown that stronger land tenure support by the state is correlated with slowed deforestation (Angelsen and Kaimowitz, 1999).

### *Institutions and governance*

Institutional factors such as governance and political instability contribute to deforestation in a variety of contexts. The structure of property rights, environmental laws, and decision making systems are all important aspects of government that influence which groups are granted concessions or allowed to extract natural resources. Governments also have an enforcement responsibility. Due to corruption and lack of regulatory enforcement, many countries with significant tropical forest resources are unable to monitor and prevent deforestation in areas where it is illegal (Lambin et al., 2003). Frequently, protected areas in certain countries are subject to illegal logging simply due to lack of enforcement. Researchers found that in Gunung Palung National Park in West Kalimantan, Indonesia, approximately 38% of the lowland forests were illegally deforested in a 14 year span (Curran et al., 2004).

Over the past several decades, developing nations have increasingly adopted decentralization policies as a strategy to improve governance, local empowerment, and management of resources (Tacconi, 2007). A study commissioned by the World Bank found that over 80% of developing countries with populations greater than five million were attempting to decentralize their governance structures (Silver, 2003). Donor agencies and development organizations such as the World Bank and International Monetary Fund espouse decentralization as a means to increasing accountability, transparency, and democracy in developing countries (McCarthy, 2004).

The popularity of decentralization policies among key donors agencies and academic theorists has resulted in attempts by many developing countries with significant forest resources to transfer power over forest resources from central to local governments. While, in theory, local control over resources leads to improved resource governance, in practice, decentralization has led to power struggles over resources and confusion over delegation of powers in countries like Indonesia (Ribot et al., 2006; Thorburn, 2002). Rhetoric surrounding decentralization of natural

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There are often high rates of deforestation in countries where clearing the land is the primary mechanism for claiming property rights.

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resource management supports the idea that local government control will lead to a scaling up of community-based natural resource management and more sustainable forestry practices in countries like Indonesia (McCarthy, 2004). Examining the impacts of decentralization on Indonesia's forests reveals a significantly different outcome across much of the country. Once decentralization was put into place and local districts were allowed to grant small forest concessions, the result in some areas was a rapid harvest of remaining lowland forest (Curran et al., 2004). Despite the belief that decentralization would lead to more sustainable community-based methods of management, many communities capitalized on having greater control over their natural resources by cutting down more of the forest (Thorburn, 2002).

### **Economic factors**

#### *International trade and economic integration*

International trade, as well as the push for economic liberalization and integration, has also shaped land use trends. Economic liberalization policies, such as the institution of free trade and the removal of tariffs and trade barriers, have typically encouraged incremental agricultural land conversion. These policies can change capital flows and investments in a region, leading to land use changes which may include deforestation (Lambin et al., 2003). As governments continue to remove barriers to trade and focus on export markets, individuals become increasingly driven by market price fluctuations. In this way, conversion of land to agriculture becomes more closely correlated to global commodity markets. Governments have also been influenced by the IMF to institute structural adjustment programs that can change agricultural practices by removing price supports, subsidies, and barriers to trade (Roebeling and Ruben, 2001). This may or may not increase pressures to convert land conversion for agriculture depending on current commodity prices and economic cycles. In sum, the net impact of economic liberalization is not clear. On the one hand, economic liberalization can increase investment in industrial agriculture, leading to higher levels of deforestation and land degradation. On the other hand, economic liberalization can increase productivity and drive the implementation of more environmentally-sustainable agricultural technologies. With the right incentives, it may also encourage participation in alternative markets that support improved environmental practices through eco-labeling and green certification systems (Lambin et al., 2003).

#### *National economic policy*

National economic policies are largely driven by the need for economic growth and national security and do not always consider the resulting impacts these policies have on the forest (Naughton-Treves, 2004). Depending on the region, economic policies driving deforestation for agriculture include credit policies, subsidies for agricultural inputs and outputs, taxation schemes, and agricultural price supports (Naughton-Treves, 2004; Martin, 2008). Currency devaluation has also been found to correlate with deforestation for agriculture as it encourages individuals to increase agricultural

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While, in theory, local control over resources leads to improved resource governance, in practice, decentralization has led to power struggles over resources and confusion over delegation of powers in countries like Indonesia.

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production in order to compensate for economic insecurity (Mertens et al., 2000; Richards, 2000). Not surprisingly, when dollar-denominated, global commodity prices are high and the cost of local farm inputs are steady or decreasing, deforestation generally increases (Chomitz et al., 2007). While commodity prices are most directly affected by subsidies, currency devaluation, exchange rates, and international trade, farm input prices vary most significantly in response to credit access and subsidies (Chomitz et al., 2007). In an economic simulation for Costa Rica, a 20% increase in input price subsidies resulted in a 2% decline in forested area (Roebing and Ruben, 2001). Similarly, a 20% increase in the availability of formal credit also led to a 2% decrease in forestland. Government subsidies and access to credit for farm equipment can lead to mechanization and intensification of agricultural production, lowering overall costs and further driving land conversion to agriculture (Azevedo-Ramos, 2007). In this way, national economic policies can create unintended and perverse incentives to deforest land for agriculture.

The interplay between international commodity markets and national economic policies can result in deforestation for agricultural uses. In the Brazilian Amazon, a combination of government incentives for forest conversion coinciding with an increase in beef prices led to the conversion of millions of hectares to low-productivity pasture lands (Chomitz et al., 2007; Azevedo-Ramos, 2007). In Cameroon, when cocoa and coffee prices began to decline in 1985 and the country entered an economic crisis, the government increased subsidies for agricultural inputs, leading to the expansion of agricultural cultivation into forested lands (Mertens et al., 2000). National economic policies have supported – and have a great potential to support – high-productivity agriculture that can reduce the need for agricultural acreage from forestland. Unfortunately, in several regions, national economic policies to promote economic growth have in fact led to the expansion of agriculture at the expense of forests.

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### *Household and local economies*

At the household level, land use decisions are directly linked to local market access and fluctuations in on-farm and off-farm wages. Local market access is generally constrained by roads and transportation infrastructure. When greater market access and economic opportunities emerge, individuals will often respond by increasing production of valuable commodities and expanding agricultural operations (Lambin et al., 2003). In Cameroon, the villages with the greatest increase in access to local markets through improved food distribution networks were also found to be the villages with the highest forest loss (Mertens et al., 2000).

In terms of labor markets, decreases in on-farm wage rates have been linked to agricultural conversion, while increases in off-farm wages and employment have been associated with decreased deforestation rates (Barbier and Burgess, 2001). In Puerto Rico, when coffee prices dropped and city wages increased, there was migration to the cities, leading to decreased deforestation and forest regeneration (Chomitz et al., 2007). Remittances from family members abroad can also serve to reduce deforestation as these households feel less economic pressure to expand croplands

(Lambin et al., 2003). Thus, improved market access and decreased on-farm wage rates can encourage households to make decisions to deforest for agricultural expansion, while improved off-farm wages and opportunities can lead to decreased rates of deforestation and even forest regeneration.

### *Culture and household-level decision making*

On a day-to-day basis, individuals make land use decisions based on cultural preferences, available information, and cultural and economic expectations (Lambin et al., 2003). The aggregation of these individual decisions can translate into extensive deforestation and land use change. Properly organized and with proper incentives, they can also lead to conservation and avoided deforestation. Influenced by the political economy, biophysical characteristics of the land, and culture of the region, individuals will make a rational decision as to what type of land use they choose to implement, varying from swidden agriculture, diversified production systems, and agrosilvopastoral systems to intensive monoculture plantations and pasture (Lambin et al., 2003; Bebbington, 1996). This is important to consider when designing carbon sequestration incentives, particularly since carbon sequestration will be one of many land use choices available to landowners.

While perennial and agroforestry systems have been associated with lower rates of deforestation, pasture can contribute to higher deforestation rates (Lambin et al., 2003). In the case of the Atlantic Forest, which extends into Argentina, Paraguay, and Brazil, only 7.5% of the primary vegetation is still intact due to land use change (Myers et al., 2000). In the province of Misiones, Argentina, high rates of deforestation of the Atlantic Forest are the result of national agricultural and economic policies, as well as the increased use of mechanized agricultural production methods. This has not only resulted in loss of forestland, but has led to increased monoculture agricultural and forestry plantations (Carrere, 2005; Lawson, 2009). The regional political and economic context, combined with cultural preferences, affect the decision to adopt a particular agricultural system and its management techniques, which can have positive or negative effects on forestland acreage.

## **THE ROLE OF CLIMATE POLICY IN REDUCING TROPICAL DEFORESTATION**

One potential mechanism for addressing tropical deforestation has emerged through the international climate negotiations under the United Nations Framework Convention on Climate Change (UNFCCC). Policy incentives to reduce deforestation and forest degradation, or REDD, are being considered as part of a new climate agreement to be negotiated at the next Conference of the Parties in December 2009. There are many issues that must be taken into account when designing policies to protect forests either using a fund or a carbon market. Since national and local governments would ultimately administer domestic REDD programs, implementation challenges in developing countries must be taken into account when allocating funds for REDD. For governments that have weak regulatory enforcement

structures, it is difficult to monitor and enforce behavior that maintains the carbon stock of standing forests. Similarly, for governments where corruption is an issue, it may be difficult to ensure that REDD funding is equitably distributed to individuals who are reducing deforestation on their lands.

The issues of land tenure and economic inequalities come to the forefront in establishing institutional capacity for REDD. For farmers who do not have title to their land, there must be other incentive structures to promote forest and agricultural management for carbon sequestration. It is unclear today whether farmers will have access to REDD funding if they lack ownership of the land. One solution might be to promote existing cooperatives and farmers' associations to channel REDD funds to smallholders who keep their land forested. Cooperatives, farmers' associations, and extension agencies could also serve as a mechanism to provide training on REDD and assist smallholders in obtaining payments to support reduced deforestation. The development of effective institutions at both a local and national level is key to promoting reduced deforestation and emissions from land use change and allowing equitable access to payments for REDD.

For small landholders, deforestation is often the only solution to support their families and their livelihoods. Thus, in order to stem deforestation for agriculture, there must be sustainable economic development, as well as adequate education and health services provided to rural communities. It is essential for policymakers to take a holistic view of the complex factors that work to drive deforestation. By examining and understanding this complexity, it is clear that REDD policies are only part of the solution to reduce deforestation and promote carbon sequestration. What is required is a combination of policies and market mechanisms that simultaneously promote sustainable economic growth and reduce poverty and economic inequalities, while protecting forests from further clearing for agriculture.

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## CONCLUSIONS AND POLICY RECOMMENDATIONS

There is no single model to explain economic drivers of tropical deforestation across all regions and scenarios. The circumstances that drive deforestation are locally based and depend upon a variety of factors that include social, political and geographical considerations. A comprehensive look at these drivers requires a multi-scale analysis that addresses how these factors interact. For example, at a local scale, population pressure and poverty can be shown to lead to deforestation, but these explanations are limited in their ability to describe the scale of deforestation that many tropical countries have experienced in recent years. Policies to address the drivers of deforestation must therefore be multidimensional and examine the underlying causes of socioeconomic factors along with larger macroeconomic policies and institutional arrangements that may affect local level land use decisions.

As REDD negotiations continue to consider the various ways carbon financing can be used to help preserve carbon stored in standing tropical forests, it is important to consider what are generally accepted economic drivers of tropical deforestation, alongside what is less well understood:

- Significant drivers of deforestation are frequently context-specific and are affected by local political, socioeconomic, cultural, and biophysical factors.
- The roles of population growth and poverty in driving deforestation have often been overstated for certain regions (Africa may be an exception).
- Transportation infrastructure is strongly correlated with deforestation; therefore, supporting national policies that reduce development pressure on forests or require improved land use planning could be an effective method for reducing deforestation along roads.
- Fluctuating commodity prices for agricultural crops, timber, and livestock can directly affect household decision-making to deforest for agriculture or to maintain the forest.
- Economic policies at the national level – including subsidies and access to credit – can play a key role in influencing deforestation for agriculture.
- Agricultural technologies that improve productivity, save capital, and create jobs may not necessarily increase deforestation pressure. Agricultural technologies that increase yields, are capital intensive, and allow farmers to employ less labor in fact may exert stronger pressure on individuals to deforest.
- The complex interaction between drivers of deforestation at different scales suggests that no single policy can be effective in slowing or halting deforestation, even with a REDD scheme.
- Regional models of deforestation drivers must account for heterogeneity across landscapes and regions as well as the complexity of interacting drivers.
- It is unknown how these drivers will continue to shift over time since demographic trends, institutional factors, and economic policies are constantly changing.
- In order to have a successful REDD mechanism, it will be essential to address many of these underlying drivers of deforestation in tropical regions. REDD should provide incentives or contain eligibility criteria for countries seeking REDD money to start undertaking some of these broader economic and governance reforms.

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## Chapter 15

# Large and Intact Forests: Drivers and Inhibitors of Deforestation and Degradation

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### EXECUTIVE SUMMARY

This chapter examines the political, economic, geographic, and biophysical reasons for the presence of the remaining large and intact forests of the world. It discusses why these forests remain relatively undisturbed, and analyzes current drivers of deforestation and degradation. It concludes with recommendations for policymakers to help incorporate these forests and their carbon stocks into initiatives designed to mitigate the damaging effects of global climate change.

A review of published literature on this topic revealed the following general trends:

- Although clearing of forestland continues at high rates in many parts of the world, large tracts of continuous, intact forest still cover roughly a quarter of originally forested biomes. These forests are unique in that they represent stable, yet vulnerable, carbon stocks. Because of both their extent and the large amount of carbon stored within these forests, their protection must be a significant part of any global policy initiatives to combat climate change.
- Presently, the vast majority of the world's remaining tracts of continuous, intact forest are concentrated within continental interior boreal and tropical wet and semi-evergreen forest biomes. Within the boreal biome, these forests cover the northern and largely inaccessible regions of Canada, Alaska, and Russia. Within the tropics, vast wet and semi-evergreen forests are found within the Amazon Basin of South America and the Congo Basin of central Africa.

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- In terms of carbon storage, three of the countries with the largest area of remaining intact forestland (Brazil, Russia and the Democratic Republic of Congo) hold 384 billion tons of carbon dioxide equivalents in above and below ground biomass, both dead and living. For comparison, global emissions from energy consumption were estimated at 29 billion tons of carbon dioxide in 2006.
- In addition to the important role these forests play in the global carbon cycle, their protection from land conversion yields highly significant co-benefits. Evidence suggests that large, intact forests have significant cooling effects on both regional and global climates through the accumulation of clouds from forest evapo-transpiration, which also recycles water and contributes to the region's precipitation.
- The low fertility and high vulnerability of the soils in interior regions of the tropics has slowed the development of permanent agriculture in these areas. Human communities that reside in these regions typically have low population densities and rely on hunting and migratory cultivation.
- Colonial history played a role in low human population densities of the large forest interiors of South America and Central Africa by being primarily resource-driven, resulting in less permanent European settlements outside of administrative extractive hubs. In addition, after settlers were established (largely on the coast), European diseases decimated native populations even in areas largely untouched by European settlers.
- The geography of remoteness is of critical importance in explaining why intact forests exist where they do, namely, in continental interiors. Much of the world's population is concentrated within 100 km of coasts, with population density decreasing as one moves to the interior. In addition, large mountain ranges (e.g. the Andes) and rugged topography (e.g. New Guinea) serve as barriers.
- A shared trait among the world's large and intact forests is a lack of foreign investment; however, globalization of markets and export products/crops have facilitated forest exploitation and land conversion of large intact forests in recent years.
- Lack of government presence has resulted in poor infrastructure development, few government services, and an inability to integrate these regions into larger market and governmental/organizational structures.
- Some countries, in order to facilitate rural in-migration to the forest frontier, in part to secure sovereignty where there are adjacent country claims and in part as a "poverty release valve," have provided agricultural subsidies, free land, and seeds to colonial settlers.
- At the country scale, forest loss often follows a Kuznets curve, whereby deforestation rates are initially static, increase during industrialization when populations are growing, and finally stabilize into an equilibrium state.

However, growing economies, increasing affluence, extreme levels of poverty, and rapid decreases in prosperity during periods of economic crisis can alter this trend and lead to unanticipated deforestation and forest degradation.

- Deforestation of large sections of the central Amazon Basin is directly attributable to governmental stimulus plans, road building programs, and subsidies for livestock production.
- The construction of roads linking both core forests and frontier forests to population centers and export markets is tied to increasing rates of deforestation. While such public highways have caused localized deforestation, the lack of parallel access outside of these roads leaves large tracts of forest intact. Unofficial roads, however, form extensive, dense networks to support transportation of the resources being harvested or extracted and can exacerbate deforestation.
- A lack of governance, coupled with the presence of infrastructure, is often a precondition for widespread illegal operations that promote deforestation (e.g. logging, illicit drug trade). However, a lack of governance with no infrastructure inhibits illegal operations that promote deforestation.

**Keywords:** Amazon, Brazil, Central Africa, colonial, Democratic Republic of Congo, development, governance, logging, New Guinea, population, roads, Russia

## INTRODUCTION

Each year, approximately 13 million hectares of tropical forest, equivalent to the land area of Greece, are felled, burned, and converted to an alternative land use (FAO, 2005). When such land is converted, carbon stored within above ground biomass, downed woody debris, and soil is released from forests to the atmosphere as carbon dioxide and methane gas. Land use change is currently responsible for approximately 20% of global greenhouse gas emissions, or 0.9 million tons of carbon dioxide equivalents per year (Defries et al., 2002).

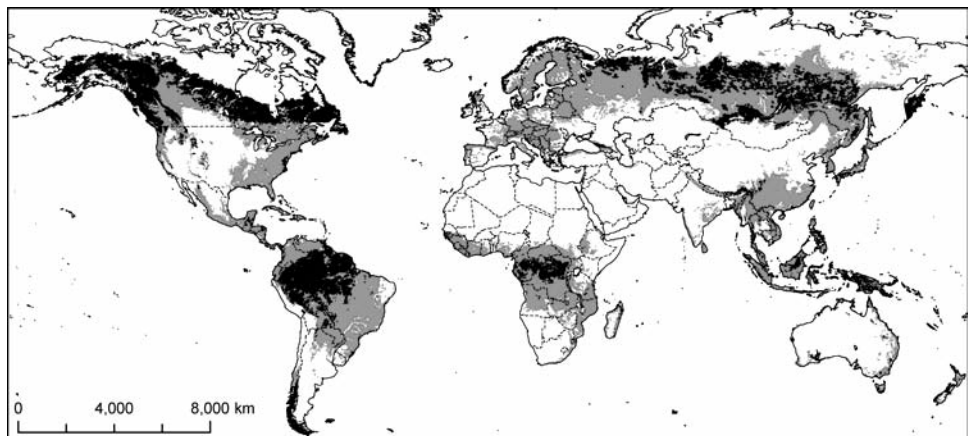
Although clearing of forestland continues at high rates in many parts of the world, large tracts of continuous, intact forest still cover roughly a quarter of originally forested biomes (Potapov et al., 2008). These forests are unique in that they represent stable, yet vulnerable, carbon sinks. Because of both the aerial extent and significant amount of carbon stored within the world's remaining large and intact forests, their protection must be part of any global policy initiatives to combat climate change. This chapter will examine the world's remaining large and intact forests. It will discuss the political, economic, geographic and biophysical reasons why these forests remain relatively undisturbed, and will analyze current drivers of deforestation and degradation. It will conclude with several recommendations for policymakers to help incorporate these forests and their carbon stocks into initiatives designed to mitigate the damaging effects of global climate change.

### Defining large and intact forests

Large and intact forests are defined as unbroken expanses of forest with negligible levels of human-induced degradation, resource exploitation, and fragmentation. As part of this definition, there is a continuous spatial threshold that these forests must meet (~100,000 km<sup>2</sup>). Intact forests are functioning ecosystems characterized by full species assemblages, naturally occurring disturbance regimes, and unaltered hydrological patterns. They are distinct from exploited and/or degraded forests, which tend to occur as patches within a mosaic of developed and agricultural areas (Chomitz, 2006). It is important to distinguish these degraded-mosaic forests from intact forests. While secondary, mosaic, and degraded forests play important roles for biodiversity, social values, carbon sequestration, and climate change amelioration, they are functionally distinct from large and intact forests in ecological terms, disturbance regimes, and management objectives. Within large and intact forests, deforestation primarily occurs along agricultural frontiers and generally does not occur from within the core interior forested areas (Chomitz, 2006). Because large intact forests have high area-to-perimeter ratios, they have fewer access points than fragmented, mosaic forests, which helps to shield the interior areas, at least in part, from deforestation.

Presently, the vast majority of the world's remaining tracts of continuous, intact forest are concentrated within continental interior boreal and tropical wet and semi-evergreen forest biomes (Figure 1).

**Figure 1** Large and intact forests are highlighted in black. Other forested areas are highlighted in gray.



Source: Potapov, P., et al., 2008. Mapping the world's intact forest landscapes by remote sensing. *Ecology & Society* 13, 51.

Within the boreal biome, these forests cover the northern and largely inaccessible regions of Canada, Alaska, and Russia. They are characterized by short, cool summers followed by long, cold winters and are often dominated by single-stand coniferous forests with limited plant species diversity (Wieder and Vitt, 2006). Within the tropics, vast wet and semi-evergreen forests are found within the Amazon Basin of South America and the Congo Basin of Central Africa. The Amazon Basin (5.5

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million km<sup>2</sup>) and the adjacent forests of the Guyana Shield contain the largest intact and contiguous tropical forest in the world. This forest is shared by nine nations of South America (Bolivia, Brazil, Columbia, Ecuador, French Guiana, Guyana, Peru, Suriname and Venezuela), although the majority of this forest lies within the borders of Brazil (Table 1) (Encyclopedia Britannica, 2009).

**Table 1 Forest and carbon data for countries and regions with large intact forests. Cells shaded in grey contain incomplete data.**

	Forest Area	Primary/Intact Forest Area	Carbon in Biomass	Carbon Biomass Density	Deforestation Rate (2000-2005)	Loss of Primary Forest (2000-2005)
			Million tons CO <sub>2</sub> equivalents			
<b>Amazon Basin/ Guayanas</b>	(1000 ha)	(1000 ha)	(Tons/ha)	(Tons/ha)	(1000 ha/yr)	(1,000 ha)
<i>Bolivia</i>	58740	29360	19436	331	-270	-135.2
<i>Brazil</i>	477698	415890	181059	379	-3103	-3466
<i>Columbia</i>	60728	53062	29588	487	-47	-56.16
<i>Guayana</i>	15104	9314	6320	418	0	0
<i>Peru</i>	68742	61065	Unknown	Unknown	-94	-224.6
<i>Suriname</i>	14776	14214	20890	1414	0	0
<i>Venezuela</i>	47713	Unknown	Unknown	Unknown	-288	Unknown
Total Amazon/Guayana Nations	743501	582905	257293		-3802	-3881.96
<b>Congo Basin</b>						
<i>Cameroon</i>	21245	Unknown	6980	329	-220	Unknown
<i>CAR</i>	22755	Unknown	10280	452	-30	Unknown
<i>Congo</i>	22471	7464	19014	846	-17	-5.647
<i>DRC</i>	133610	Unknown	85045	637	-319	Unknown
<i>Equatorial Guinea</i>	1632	Unknown	423	259	-15	Unknown
<i>Gabon</i>	21775	Unknown	13370	614	-10	Unknown
Total Congo Basin Nations	223488	7464	135112	3136.188798	-611	-5.647
<b>New Guinea</b>						
<i>Papua New Guinea</i>	29437	25211	Unknown	Unknown	-139	-250.2
Total New Guinea	29437	25211	Unknown	Unknown	-139	-250.2
<b>Boreal</b>						
<i>Russian Federation</i>	808790	255470	118211	146	-96	-532.2
<i>Canada</i>	310134	165424	Unknown	Unknown	0	0
Total Boreal Nations	1118924	420894	118211	146	-96	-532.2

Source: Data compiled from FAO, 2005.

The Congo Basin in Africa (3.5 million km<sup>2</sup>) contains the world's second largest contiguous tropical forest and is largely located within the borders of six nations (Cameroon, Central African Republic, Congo, the Democratic Republic of the Congo, Equatorial Guinea, and Gabon) (CARPE, 2001). Similar to the Amazon Basin, a single country (the Democratic Republic of the Congo (DRC)) contains a majority of the region's forested area (Table 1). Smaller, but significant, intact tropical forests are found on the islands of Sumatra, Borneo, and New Guinea, the highlands of mainland Southeast Asia, and the Atlantic coast of Central America. This chapter will focus on the boreal forests of Canada and Russia (including temperate forests bordering these boreal forests), the Amazon and Congo Basins, and New Guinea.<sup>1</sup> It will not consider forests in temperate regions, with the exception of temperate forests

<sup>1</sup> Because of the difficulty in segregating countrywide data in Indonesia from data specific to New Guinea (Irian Jaya), discussions regarding New Guinea will be focused on the Papua New Guinea half of the island.

bordering boreal forests in Canada and Russia, since most forest cover in temperate parts of the world is patchy and dominated by secondary growth (Figure 1) (Potapov et al., 2008).

A significant proportion of large and intact forests are within the borders of a small number of countries. For example, Brazil, Canada, and Russia contain 63.8% of the area of the world's remaining large and intact forests within their borders (Potapov et al., 2008) (Figure 1).

### WHY IS PROTECTING LARGE AND INTACT FORESTS SO IMPORTANT?

Large and intact forests are extremely important for the multitude of ecosystem services they provide. Despite their global importance, however, only 18% of these forests had been designated as Protected Areas<sup>2</sup> as of 2008 (Potapov et al., 2008). Unfortunately, even with this designation, protection is minimal. While these forests have remained largely intact, they are often in areas under increasing pressure from land use conversion, road building, and timber extraction. As one example, recent history has seen rapid, large-scale deforestation in Borneo due to illegal logging and industrial-scale land conversion for agriculture (Curran et al., 2004). With increasing rates of deforestation, and its impact on global greenhouse gas emissions, there is broad consensus that continued illegal logging and aggressive industrial land conversion practices must be addressed immediately, either through market-based incentives such as carbon credits, regulatory structures to improve governance, or a combination of both (Zhang et al., 2006; Betts et al., 2008; Buchanan et al., 2008; Nepstad et al., 2008).

<sup>2</sup> A Protected Area is defined as any land that is given protected status as an extractive reserve, national park, indigenous reserve or wildlife reserve.

<sup>3</sup> For more information regarding carbon sequestration and storage, see the science chapters of this volume.

<sup>4</sup> While the total amount of carbon stored is highest in primary or intact forests, rates of carbon sequestration are highest in fast-growing secondary forest. This is why large and intact forests are considered carbon "reservoirs" as opposed to the carbon "sinks" of growing secondary forests.

### Carbon sequestration and storage<sup>3</sup>

Carbon markets may provide effective financial incentives to deter land conversion and illegal logging in large and intact forests simply due to the sheer amount of carbon stored in these forested areas. In terms of carbon storage, three of the four countries with the largest area of remaining intact forestland (Brazil, Russia, and the Democratic Republic of Congo) hold 384 billion tons of carbon dioxide equivalents in above and below ground biomass, including dead and living biomass (FAO, 2005) (Table 1). In comparison, global emissions from energy consumption were estimated at 29 billion tons of carbon dioxide in 2006 (EIA, 2006). The significant amount of carbon stored within these countries is due to the fact that primary or intact forests contain higher densities of carbon in soils and living biomass than degraded or secondary forests (Olson et al., 1985).<sup>4</sup>

### Co-benefits of protecting large and intact forests

In addition to the important role these forests play in the global carbon cycle, their protection from land conversion yields highly significant co-benefits as well.

First, large intact forests have been shown to play a role in regional climate regulation (Hoffman et al., 2003; Spracklen et al., 2008). In the boreal region, intact forests have a

significant cooling effect on both regional and global climates through the accumulation of clouds from boreal forest evapo-transpiration (Spracklen et al., 2008). The cooling effect that large forests exert via evapo-transpiration has also been demonstrated in the tropics, particularly in the Amazon. A large portion of the precipitation in interior/continental regions of the Amazon Basin is derived from evapo-transpiration that is released over the course of a day (Makarieva and Gorshkov, 2007). When there is deforestation of forest frontiers or edges, interior regions of wet tropical forests often cannot sustain their current forest type due to changes in precipitation patterns (Makarieva and Gorshkov, 2007). When large swaths of previously intact tropical forests are cleared, evapo-transpiration occurs much more rapidly, leading to disrupted precipitation patterns downwind of the deforestation (Roy et al., 2005). In one study, a model of precipitation in the Congo Basin suggested that rainfall could be reduced by 10% in certain regions as a result of deforestation (Roy et al., 2005).

Second, when changes to precipitation patterns occur in tropical forests, they can lead to altered fire regimes, which can impact the resilience of remaining forests. Many countries in the tropics with significant rates of deforestation and land conversion now experience much more frequent and severe fires (Siegert et al., 2001; Hoffman et al., 2003). These resulting fires can exacerbate deforestation and degradation rates in remaining forests, which in turn can have a large impact on global carbon emissions (Hoffman et al., 2003). This effect was seen in the 1997 fires on the island of Borneo, which released an estimated range of 8-25 billion tons of CO<sub>2</sub> equivalent into the atmosphere, equal to 13-40% of the mean annual global emissions from fossil fuels (Page, 2002).

Third, there is ample evidence that forest fragmentation and degradation have significant effects on both floral and faunal species composition within a given region (Curran and Leighton, 2000; Hoffman et al., 2003; Roy et al., 2005). Certain changes in plant species composition can compromise the resilience of an entire ecosystem and reduce its ability to withstand disturbance. Many plant species rely on large expanses of forest for their regeneration and cannot effectively reproduce in mosaic or fragmented forests (Curran and Leighton, 2000). In addition to protecting plant biodiversity, these forests also provide some of the only remaining suitable habitat for wildlife in their respective regions (Joppa et al., 2008).

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## **COMMON FEATURES OF THE WORLD'S REMAINING LARGE AND INTACT FORESTS**

Today's large and intact forests share a number of common traits that have historically hindered deforestation. Many of these forests also, not surprisingly, share similar risks of potential deforestation. There are, however, regional variations which are important to keep in mind. For example, while industrial-scale agriculture and regional infrastructure play a strong role in deforestation in the Amazon, they are not considered significant threats to the forests of the Congo Basin. Reasons for regional differences are complex and often due to both local and international factors. Common features as well as regional differences are summarized in Table 2.

**Table 2 A comparison among large intact forest regions of key factors facilitating persistence, and current drivers of deforestation and degradation.** A single X denotes regionally important factors and XX denotes highly important factors.

	Amazon Basin/ Guyanas	Congo Basin	Boreal Forests	New Guinea
<b>Key Historical Factors Allowing Forest Persistence</b>				
Biophysical Limitations				
<i>Soil Infertility</i>	X	X	X	
<i>Climatic Barriers to Agriculture</i>			XX	
Low Population Density	X	X	XX	
Biogeographical Isolation				
<i>Inaccessibility to Markets</i>	XX	XX	XX	X
Governmental Factors				
<i>Lack of Governmental Capacity</i>	X	XX		X
<i>Lack of infrastructure</i>	XX	XX	X	X
Low Levels of Foreign Investment	X	XX	X	X
<b>Current Drivers of Deforestation</b>				
Poverty				
<i>Subsistence Extraction and Agriculture</i>	X	X		
Governance				
<i>Land Tenure Insecurity</i>	X	X		X
<i>Poorly Designed Concession Systems</i>				
<i>Corruption</i>	X			X
<i>Illegal Resource Extraction</i>				
Infrastructure Expansion	XX	X		
International Trade and Investment				
<i>Poorly Managed Timber Extraction</i>				
<i>Foreign Investment</i>	X	X		X
<b>Current Drivers of degradation</b>				
Poverty				
<i>Subsistence Extraction and Agriculture</i>	X	XX		X
Governance				
<i>Land Tenure Insecurity</i>	X	XX		X
<i>Poorly Designed Concession Systems</i>	X	X	X	X
<i>Corruption</i>	X	X	X	
<i>Illegal Resource Extraction</i>	X	X	X	X
Infrastructure Expansion	XX	XX		
International Trade and Investment				
<i>Poorly Managed Timber Extraction</i>	X	X	X	X
<i>Foreign Investment</i>	X	X		X

### Why have these forests remained intact?

Deforestation rates in these forests are often much lower than other regions where land use conversion continues at a rapid pace. For example, in the Congo Basin nations of the Central African Republic, Congo, Democratic Republic of the Congo, and Gabon, average annual deforestation rates between 2000 and 2005 were only



0.13%, 0.076%, 0.24% and 0.046% respectively of their total forest area per year (Table 1) (FAO, 2005). In contrast, the deforestation rate in Indonesia and Cambodia was 2.0% per year from 2000 to 2005 (FAO, 2005). The remaining large and intact forests of the world persist to this day because of biophysical, biogeographical, demographic, governmental and economic factors that have allowed these forests to remain relatively undisturbed (Table 2), while primary forests in other parts of the world have gradually decreased in size and extent. An historical understanding of why deforestation rates in these areas have remained low will shed some light on the risks these forests may face as conditions change.

## Biophysical limitations

### *Tropics*

The geography of human settlement is neither random nor uniform. Although the wet tropical rainforests of the world support an ecosystem of tremendous biodiversity, they are typically an inhospitable place for humans to live. The term “Counterfeit Paradise” has been coined to describe this paradox between biological richness and the physical impoverishment of many tropical forest dwellers (Meggers, 1995). The majority of soils within the Congo Basin and Upper Amazon are classified as oxisols by the U.S. Department of Agriculture (Natural Resources Conservation Service, 2005). These soils are characterized by extremely low levels of fertility, small nutrient reserves, low cation exchange capacities, and shallow organic layers. Because the nutrients in oxisols are rapidly leached by rainfall and because tropical forests receive extremely high amounts of precipitation, these forests must undertake rapid decomposition and nutrient cycling to prevent nutrient depletion (Markewitz et al., 2004). As a result, when they are converted to agriculture, these soils are typically only productive for a few years (Montagnini and Jordan, 2005).

The low fertility and high vulnerability of the soils in interior regions of the tropics have prevented the development of permanent agriculture and led to cultures with low population densities which rely on hunting and migratory cultivation. These shifting cultivation/swidden cultures often do not put too much pressure on forest resources, which has helped to preserve large and intact forests in many of the areas they inhabit (Dove, 1983).

While many of the interior regions of tropical areas have low soil fertility, other tropical areas can be highly suitable for agriculture. The volcanic, highly fertile soils of Java and the Great Lakes Region of Africa support some of the highest rural population densities in the world despite being located in areas that are classified as tropical rainforest (Natural Resources Conservation Service/USDA, 2000). As a result of soil fertility and the ability to support large populations, the forests in these regions were largely converted to alternative land uses centuries ago. The relatively fertile highlands of New Guinea, one of the focal areas of this chapter, are an exception. This is mostly due to the fact that the soils of New Guinea are typically inceptisols, which although suitable for agriculture, are highly erodable on steep slopes, making agriculture logistically difficult (Natural Resources Conservation Service, 2005).

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Local climate may also play a role in deterring widespread agriculture within tropical basins. In the Brazilian Amazon, low levels of precipitation were shown to be the most important determining factor influencing the deforestation of land for agriculture and pasture. In fact, precipitation levels were found to be more important than access, soil fertility, and land protection status (Chomitz and Thomas, 2003).

### *Boreal*

Agriculture within the boreal ecosystems is inhibited by both poor soil quality and a climate that is unsuitable for most agriculture. Winters in the boreal zone are both long and extremely cold. Spring cold snaps and short growing seasons make agriculture in boreal forest regions unprofitable and unlikely to provide sufficient nourishment of large human settlements, particularly given seasonal risk (Wieder and Vitt, 2006). Moreover, many boreal soils are classified as spodosols or gellisols. Spodosols tend to be acidic, have poor drainage, and low fertility while gellisols typically contain permafrost within two meters of the soil surface (Natural Resources Conservation Service, 2005). This makes it nearly impossible to undertake successful agricultural activities.

## **Population density**

### *Population patterns*

One of the more obvious shared traits among large and intact forests is that they are found where human populations are low. While low population densities are largely a result of the biophysical limitations of these regions, they are also due to biogeographical isolation and historical factors. The Amazon Basin, Congo Basin and boreal forests of North America and Eurasia all have population densities of less than 10 people per km<sup>2</sup> (Natural Resources Conservation Service/USDA, 2000). Within these regions, rural population density is often much lower. For example, the population density in rural areas of the Peruvian Amazon was calculated to be about 1.6 people per km<sup>2</sup>, in an area the size of roughly 715,000 km<sup>2</sup> (Instituto Nacional de Estadísticas y Informática, 2007). In many of our focal region nations, populations are highly urbanized and only a relatively small proportion of their populations live in rural areas. Notable exceptions to this are the DRC and Papua New Guinea, which both have largely rural populations (Table 3).

### *Colonial history in the Amazon Basin*

In the Upper Amazon and Guyana Shield of South America, colonial history has played a large role in the low population densities of the interior portions of these countries. In tropical South America, for example, colonization was much more resource-driven, resulting in less permanent European settlements outside of administrative extractive hubs. In addition, after settlers were established (largely on the coast), European diseases decimated native populations even in areas largely untouched by European settlers (Diamond, 1997). This assertion is supported by ongoing archeological research which indicates that pre-colonial indigenous populations within the Amazonian basin were significantly larger and more

urbanized than those encountered after colonists arrived (Mann, 2000). In contrast, the colonization of North America fits the “deep settler” model, in which Europe sent large numbers of immigrant families to settle permanently in the New World (Wolfe, 1999). This led to increased fragmentation of forested landscapes from the onset of colonization.

**Table 3 Data for some current drivers of deforestation and forest degradation in the large and intact forests**

	2004 per capita GDP <sup>c</sup>	2004 Population Density <sup>c</sup>	Population Growth Rate <sup>c</sup>	2004 Rural Population <sup>c</sup>	Growing Stock Removed <sup>c</sup>	Road Density <sup>b</sup>	Political Stability <sup>a</sup>	Control of Corruption <sup>a</sup>
	(\$US)	(Pop/km <sup>2</sup> )	(Annual %)	(% of Population)	(% harvested)	(km road/km <sup>2</sup> area)	(Range from -2.5 to 2.5)	(Range from -2.5 to 2.5)
<b>Amazon Basin/ Guayana Shield</b>								
Bolivia	1,036	8.3	1.9	36.1	0	0.07	-0.99	-0.49
Brazil	3,675	21.1	1.2	16.4	0.4	0.1	-0.22	-0.24
Columbia	2,069	43.6	1.6	23.1	nd	0.4	-1.65	-0.28
Guayana	962	3.9	0.4	62	nd	0.03	-0.32	-0.64
Peru	2,207	21.5	1.5	25.8	nd	0.03	-0.83	-0.38
Suriname	2,388	2.8	1.1	23.4	0	0.1	0.23	-0.26
Venezuela	4,575	29.6	1.8	12.1	nd	0.04	-1.23	-1.04
Avg Amazon/ Guayana Nations	2,416	18.7	1.4	28.4	0.1	0.11	-0.72	-0.48
<b>Congo Basin</b>								
Cameroon	651	35.2	1.9	47.9	1.5	0.03	-0.39	-0.93
CAR	232	6.3	1.7	56.8	0.1	0.06	-1.78	-0.9
Congo	956	11.3	2.6	46.1	0.1	0.04	-0.83	-1.04
DRC	89	24.2	3	67.7	0.3	0.03	-2.26	-1.27
Equatorial Guinea	3,989	18	2.4	51	0.9	0.06	-0.16	-1.37
Gabon	3,859	5.3	2.2	15.6	0.1	0.07	0.2	-0.85
Avg Congo Basin Nations	1,629	17	2.3	47.5	0.5	0.05	-0.87	-1.06
<b>New Guinea</b>								
Papua New Guinea	622	12.4	2.2	86.8	0.8	0.03	-0.76	-1.05
Avg New Guinea	622	12.4	2.2	86.8	0.8	0.03	-0.76	-1.05
<b>Boreal</b>								
Russian Fed.	2,302	8.5	-0.4	26.7	0.2	0.6	-0.75	-0.92
Canada	24,712	3.5	0.9	19.2	0.7	1.1	1.02	2.09
Avg Boreal Nations	13,507	6	0.25	23.0	0.45	0.8	0.135	0.59

Sources: a (Kaufmann et al., 2008); b (Central Intelligence Agency, 2009); c – (FAO, 2005)

As a result of its particular colonial legacy and the spatial-demographic patterns that resulted, population densities within the Amazon Basin in countries of Upper Amazonia (Peru, Bolivia, Ecuador, Columbia) and the Guyana Shield (Suriname, Guyana) have been much lower than those within the coastal and the Andean regions of South America. The 2005 national census found 75% of the Peruvian population to be urban dwelling, with the majority concentrated in coastal cities such as Lima, Trujillo, and Chiclayo (Instituto Nacional de Estadísticas y Informática, 2007). In many ways, the legacy of colonization is still seen in the population dynamics of the Amazon Basin today.

### *Population growth*

Despite having historically small populations, some regions with large and intact forests are experiencing rapid population growth. In Africa, the Democratic Republic

of the Congo, Congo, and Gabon have estimated population growth rates of 3.0%, 2.6% and 2.2% respectively (Table 3). Papua New Guinea, another nation containing significant large and intact forests, has a population growth rate of 2.2%. This suggests that while low population densities have historically inhibited deforestation and forest degradation in these regions, population may soon become a major deforestation and degradation driver. Meanwhile, Russia, on the other hand, is undergoing a 0.5% per year population decline.

Historically, larger populations have had higher rates of deforestation and forest degradation due to the need to support more people. Today, however, with increasingly globalized markets, local population growth may not play as significant a role in deforestation and forest degradation as one might imagine. As societies and economic trade become more global, populations growing in one region of the world can have large impacts on deforestation and forest degradation in another part of the world. One example is in the Russian Far East, an area of extremely low population density and growth, whose forests are rapidly being degraded as a result of China's economic and demographic expansion (World Wildlife Fund Forest Programme, 2007).

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### **Biogeographical isolation**

The geography of remoteness is of critical importance in explaining why intact forests exist where they do and are not found around the periphery of New York or Shanghai. Much of the world's population is concentrated within 100 km of a coast, with population density decreasing as one moves to the interior (Small and Nicholls, 2003). The Upper Amazonian regions of Peru, Colombia, Ecuador and Brazil are roughly 3,000 km from the Brazilian city of São Paulo. Although some of these areas are less than 300 km from the coast, the Andes Mountains, which span the length of the South American continent, create an effective natural barrier isolating large parts of the Upper Amazon from urban centers along coastal and intermountain population centers throughout the Andean region. This isolation has prevented the integration of the interior regions of the Amazon into regional and global markets, kept population densities low, and minimized rates of deforestation and forest degradation (Nepstad et al., 2008). Similarly, the large and intact forests of boreal Russia and North America, the Congo Basin, and New Guinea are also largely found in interior regions that have low accessibility to coastal regions (Figure 1).

Geographical isolation also restricts the connection of these areas to natural resource markets. Many studies have examined the impact of distance to market on rates of deforestation and forest degradation. These studies have almost uniformly found that areas with longer travel times to market tend to have low rates of deforestation and forest exploitation (Chomitz and Gray, 1996; Chomitz, 2006). This subject is investigated in greater depth in the "Current Drivers" section in the discussion on roads and access.

### **Lack of governance**

One legacy of geographical isolation and low population densities is political isolation from centralized national governments. The lack of government presence

within core forest regions has resulted in a lack of infrastructure development and government services, and an inability to integrate these regions into larger market and governmental structures. This has helped to maintain low levels of deforestation and forest degradation (Kaimowitz, 1997). In the Amazon Basin nations, distrust between the largely indigenous inhabitants and representatives of the national governments, who tend to be of European descent, has resulted in very low governmental capacity and integration in the Amazon region. In the Congo Basin generally, and within the DRC in particular, armed conflict, ethnic tensions, and governmental instability have generally prevented large scale forest degradation and exploitation by discouraging investment of capital (Glew and Hudson, 2007). Large sections of the upper Amazon within Colombia and neighboring Venezuela are violent and largely ungoverned due to the presence of the FARC, paramilitary groups, and large-scale cocaine trafficking. Isolated parts of the Peruvian Amazon have little government presence and are controlled by drug traffickers as well as remnants of the Shining Path guerilla group. While there are no studies linking war and conflict directly to lower deforestation rates, it is likely that their presence inhibits investment in roads, health care, and resource extraction, thus keeping overall land use conversion rates at low levels.

### **Low levels of foreign investment**

Another shared trait among many of the world's large and intact forests is a lack of foreign investment. Foreign investment can be a highly significant driver of deforestation and forest degradation, particularly through infrastructure development and natural resource extraction (Chomitz and Gray, 1996; Carr et al., 2005). In many cases, foreign investment can be a catalyst for resource exploitation by giving projects sufficient capital to overcome high initial costs of resource extraction, turning an unprofitable endeavor into one that is economically viable.

A lack of project financing is often cited as a key constraint on logging expansion, particularly in areas such as the Congo Basin (Perez et al., 2006). There are many reasons why foreign investors may be less inclined to invest in resource extraction in certain forested regions. In the Congo Basin, it is likely the result of the high risks posed by violent armed conflicts and blatant corruption (Perez et al., 2006; Glew and Hudson, 2007). By contrast, in the Amazon Basin, it is more likely driven by a lack of pre-existing infrastructure in the region (due to limited governmental capacity) and the absence of technology to make agricultural operations profitable. In recent years, however, with new technologies for soy cultivation in the southern Amazon, foreign investment, and consequently deforestation, in the region have accelerated (Wilcox, 2008).

### **WHAT CURRENTLY DRIVES DEFORESTATION AND DEGRADATION OF LARGE AND INTACT FORESTS?**

There are a number of signs that, despite the lack of historical deforestation and degradation, many regions with large and intact forests may be at risk in the near

future. For example, some researchers believe that forests within the Congo Basin will fragment into three distinct and diminished forest blocks based on models predicting future population growth, road densities, and logging concessions (Zhang et al., 2006). Two of the blocks will be east of the Congo River in the Democratic Republic of Congo, with small patches remaining around the edges of the basin. Nepstad et al. (2008) have predicted that by 2050 the Amazon rainforest could be reduced to 51% of its initial extent due to a positive feedback mechanism from fires, land use conversion, and climate change. There are several interdependent factors driving the conversion and degradation of large and intact forests, including conflict, infrastructure expansion, unclear land tenure, poor governance, and global commodity flows. Each of these drivers will be discussed separately, however it is important to note that many of these factors are related and work in tandem to drive deforestation rates.

### **Poverty, affluence, and the Kuznets Curve**

At the country scale, forest loss often follows a Kuznets curve, whereby deforestation rates are initially static, increase during industrialization when populations are growing, and finally stabilize into an equilibrium state (Ehrhardt-Martinez et al., 2002). Users of this model often draw three conclusions from this trend.

First, they conclude that growing economies are most likely to exhibit rapid deforestation. This is fairly self-explanatory, as growing economies tend to increase their use of internal natural resources both to fund domestic economic growth and to participate in export markets.

Second, they conclude that increasing affluence during economic development accelerates the rates of deforestation and degradation. Studies have shown positive correlations between rising incomes, increasing agricultural exports, and forest degradation (Barbier et al., 2005; Carr et al., 2005). When agricultural operations are largely for domestic consumption, however, they do not tend to have the same impact on deforestation. For example, within the tropics, traditional shifting agriculture is responsible for only 6% of observed land use change, and only 26% of tropical deforestation is the result of small scale agriculture as a whole (Barbier et al., 2005, Martin, 2008). Although the Kuznets curve suggests that deforestation and degradation increase with affluence during national development, it also suggests that once above a certain threshold, increasing affluence has a reverse effect (Ehrhardt-Martinez et al., 2002). This is likely due to the fact that as economies develop, their economic base becomes more diversified, with less reliance on natural resource commodities. At the same time, rising affluence tends to drive increased urbanization, which shifts populations from a decentralized agrarian base to more centralized, denser urban areas where residents do not engage in subsistence farming.

Third, users of the Kuznets model often conclude that the poorest members of society in developing nations are often not the major drivers of deforestation and degradation (Carr et al., 2005). Research has shown that within Latin America and Southeast Asia, poverty has had very little impact on increased deforestation (Chomitz, 2006). In fact, studies within the Peruvian Amazon have shown that

poverty actively constrains deforestation because labor and equipment inputs are prohibitively expensive (Zwane, 2007). This trend is particularly evident where poor populations lack access to credit. In these cases, there are so few market and labor incentives that cultivation rarely grows beyond subsistence levels.

Nevertheless, some national governments continue to claim that subsistence forest inhabitants are driving deforestation and degradation. More often, however, it has been driven by government policies that welcome large, industrial scale conversion of land to agriculture, often for the benefit of multinational entities who pay hefty prices to governments for local access (Siegert et al., 2001; Doolittle, 2007). The idea that poor, rural subsistence farmers are the chief cause of the deforestation and degradation of large and intact forests continues to be disputed. Indigenous inhabitants of large and intact forests have developed systems of resource extraction that usually, if allowed to continue undisturbed, have small impacts on the forests they inhabit (Dove, 1983; Dugan, 2007). In much of the world, deforestation and natural resource extraction are increasingly controlled by external actors who have few ties to the forests they impact (Lambin and Geist, 2003).

One exception is in the Congo Basin. Here, extreme levels of poverty have led to unsustainable extraction of wood for fuel, a strong bushmeat trade, and the expansion of subsistence agriculture into the frontiers of intact forest. These activities have in fact been significant drivers of deforestation and forest degradation in the region (Iloweka, 2004). In the DRC, many rural populations surrounding the city centers have come to rely on the collection of fuel wood for their livelihoods (Iloweka, 2004). Sunderlin et al. (2000) examined the impact that Cameroon's economic downturn during the 1980s and 1990s had on deforestation in Cameroon's Congo Basin. As incomes decreased in the crisis, local landholders were forced to clear land to feed themselves, resulting in greatly increased rates of deforestation. This phenomenon of an economic crisis driving increased deforestation was also observed in Indonesia following the Asian financial crisis in the late 1990s. Small scale farmers significantly expanded their rubber holdings and other tree crops during the crisis, with the aim of increasing future income security (Sunderlin et al., 2001). Thus, despite the fact that in general forest loss and degradation follow a traditional Kuznets curve during economic development, extreme levels of poverty and rapid decreases in prosperity from periods of economic crisis can alter the trend and lead to unanticipated deforestation and forest degradation.

## Governance

There is wide variation in the quality of governance among the regions discussed in this chapter (Table 3). The DRC lies at one end of the governance spectrum with poor governance while countries such as Canada lie at the other end. Good governance greatly increases the likelihood that countries will manage resources sustainably and take steps to control deforestation and forest degradation. In this section we will discuss the implications for forests of a lack of governance, as well as how poorly designed governmental policies can drive deforestation and forest degradation.

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### *Problems related to lack of governance and land tenure*

The Democratic Republic of the Congo (DRC) and the Republic of the Congo are good examples of what happens to deforestation and forest degradation when there is a lack of national governance. As a result of violent conflicts in these two countries and in the neighboring country of Rwanda, there has emerged a large refugee population in the two Congo nations. At the same time, the lack of national governance has allowed many forested regions to remain controlled by rebel groups. The large refugee population in the region has led to illegal and unsustainable resource exploitation and is both the result of, and the cause of, continued armed conflict in the region (Glew and Hudson, 2007). The most common forms of illegal natural resource extraction resulting from the presence of refugees are hunting for bushmeat,<sup>5</sup> followed by illegal logging.

It is important to note, however, that political instability has both positive and negative feedbacks on deforestation. On the one hand, the illegal harvest of forest products is often used to fund continued armed conflict in the region, thus perpetuating the cycle of lack of governance, increased numbers of refugees, and increased forest degradation (Glew and Hudson, 2007). On the other hand, a lack of governance often means that national governments and foreign corporations are unable or unwilling to invest in infrastructure and resource extraction. For this reason, the DRC has a negligible deforestation rate, despite ranking as a bottom tier country in terms of corruption, government performance, and human livelihoods (Table 1, Table 3) (FAO, 2005; Central Intelligence Agency, 2009). In other words, the DRC may simply be too poorly governed to have a high net rate of deforestation.

In other regions, however, a lack of governance can be a key driver of deforestation and forest degradation. In the Amazon Basin, little government presence, alongside the presence of illicit actors, can increase localized deforestation rates and lead to the unsustainable extraction of timber. Increased rates of forest conversion within coca producing areas of Colombia and Peru have been directly linked to the traffic of cocaine in areas under the control of drug-related enterprises. The U.S. State Department has estimated that some 2.3 million hectares within the Peruvian Amazon Basin, accounting for 25% of deforestation, is directly the result of coca cultivation for cocaine (Beers, 2002).

A lack of governance is often a precondition for widespread illegal logging. Illegal logging has been shown as a primary driver of forest degradation in the Russian Far East, parts of the Congo and Amazon Basin, and Southeast Asia (Auzel et al., 2004; Curran et al., 2004; World Wildlife Fund Forest Programme, 2007). Often, the presence of illegal logging leads to significant resource loss, which may reinforce cycles of poverty and forest degradation if it is permitted to continue in an uncontrolled fashion (Auzel et al., 2004).

Landholders who have secure land tenure and confidence in the permanence of their residence are more likely to make long-term investments in their land (Chomitz, 2006). When populations have tenuous land rights and risk being legally (or forcibly) removed from their land, they have little incentive to practice sustainable land management activities. Moreover, when landholders lack assurances that land will be

<sup>5</sup> Although bushmeat hunting may not directly or initially impact the forest cover and carbon storage capacity of a forest, it has been shown to impact the floristic and faunal composition of tropical forests (Nunez-Iturri and Howe 2007).



protected from appropriation, they often practice unsustainable resource extraction that both degrades previously intact forests and contributes to continued poverty long term.

### *Problems related to poorly designed policies and land tenure regimes*

Beyond a simple lack of governance, poorly planned government policy can have a major influence on deforestation and forest degradation rates. Policies that create incentives to clear forests and build roads for industrial land-based operations are major drivers of deforestation and forest degradation in all of the regions covered in this chapter (Carr et al., 2005). For example, the deforestation of large sections of the central Brazilian Amazon is directly attributable to governmental stimulus plans, road building programs, and subsidies for livestock production (Fearnside, 2007). In order to facilitate rural in-migration to the forest frontier, the Peruvian and Brazilian governments have provided agricultural subsidies, free land, and seeds to colonial settlers (Alvarez and Naughton-Treves, 2003; Fearnside and De Alencastro Graça, 2006). Within the Colombian Amazon, vague and un-enforced land tenure laws in the 1970s helped to promote deforestation (Armenteras et al., 2006). In Peru, since access to frontier land is free, colonists may gain legal title to the land once it has been deforested and put to agricultural use (Imbernon, 1999). This has created a dynamic whereby agricultural settlers are encouraged to clear land in order to gain legal title. By contrast, comprehensive land tenure laws have been shown to incentivize good behavior. Research in the Honduran Miskito region found that properly demarcating land use tenure and assigning clear communal land rights lowered rates of agricultural expansion (Hayes, 2007).

Concession policies are a corollary to land tenure issues and often drive deforestation and forest degradation in the tropics. Often, concessions are awarded for finite periods of time that are too short to make sustainable forest management a viable enterprise. Thus, concession holders often engage in short term resource extraction practices (Barr, 2001). One way that better governance could improve natural resource management and decrease rates of deforestation and forest degradation would be to reform concession policies to encourage responsible forest management. Unfortunately, concession systems are extremely profitable for governments, which makes timber concession reform a highly contentious issue (Barbier et al., 2005). As a result, in most regions, concessions systems are designed to maximize short-term governmental profit at the expense of sustainability and the local inhabitants.

### **Roads, infrastructure expansion and regional market integration**

The construction of roads linking both core forests and frontier forests to population centers and export markets is invariably tied to increasing rates of deforestation. Econometric models have found that, within the Amazon Basin, roads directly cause local deforestation (Pfaff et al., 2007). Because roads decrease the transportation costs of labor inputs, equipment, and products, they greatly increase the economic

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feasibility of agriculture and extractive activities within affected areas. In the Congo Basin, roads provide accessibility to previously intact forested areas, allowing bushmeat extraction, illegal logging, and small land clearings (Makana and Thomas, 2006; Perez et al., 2006). Fearnside (2007) also found that because roads greatly increase land values, they can lead to both violent confrontation and to increased rates of land use conversion by colonizers seeking to exert de facto ownership of their land.

Roads in the Amazon have typically been constructed to facilitate one or more of the following: natural resource extraction, extension of government control and services, and expansion of agricultural frontiers (Fearnside, 2007). Within Amazonian Brazil and Peru, the construction of roads linking core forests and frontier forests to coastal population centers has historically been part of a concerted effort to populate and consolidate government control within the Amazon Basin (Alvarez and Naughton-Treves, 2003; Fearnside, 2007). Current road building and other infrastructural projects within the Amazon Basin are aimed at regional economic integration and the transportation of agricultural goods to export markets (Perz et al., 2008). The paving of the trans-oceanic highway is expected to link ports along Peru's Pacific coast to the Atlantic coast of Brazil and to facilitate export activities of participating countries and global markets (IIRSA, 2005).

It is unclear what effects these projects will have on deforestation rates long term. While roads have been associated with accelerated rates of deforestation, they are also seen as an essential component of economic and social development. In order to minimize deforestation, illegal land clearing, violence, and the displacement of indigenous groups along new road networks, there must be clear governance structures, enforceable land use tenure and zoning laws, and the strategic positioning of indigenous and natural reserves (Fearnside and De Alencastro Graça, 2006).

### *Official vs. unofficial roads*

Recent literature has focused on the proliferation of privately funded, unofficial road networks (Perz et al., 2005b; Perz et al., 2008). Official roads tend to stretch for hundreds of kilometers and connect interior cities to population centers outside of the forest. They also tend to be financed with public funding and through international lending channels. On the other hand, the building of unofficial roads in the Amazon and Congo Basin tends to be driven by industrial scale resource extraction projects and typically does not serve population centers. The unofficial roads tend to be constructed by private interests to suit their particular needs. Perz et al. (2008) found that while public highways have caused localized deforestation, the lack of parallel access points generally leaves large tracts of forest intact. Unofficial roads, however, form extensive, dense networks to support transportation of the resources being harvested or extracted (Pfaff et al., 2007). In addition, when large roads are paved, they often stimulate the creation of extensive unofficial interior road systems that lead to deforestation and forest fragmentation (Perz et al., 2008).

## International trade and investment

### *Global trade*

The globalization of international trade has been occurring at an unprecedented rate over the last 25 years. In many regions discussed in this chapter, particularly in the Amazon and in New Guinea, the most significant impact from globalization is an expanding agricultural sector that drives deforestation and forest degradation on the frontiers of large and intact forests. Another effect of globalization is the increased demand for timber products from these regions. Total international trade in wood and paper products has increased in value from just over 50 billion USD in 1983 to over 250 billion USD in 2005 (ACPWP, 2007). Increased demand for forest products has led to widespread forest degradation, which is often exacerbated by over-harvesting and poor logging practices.

Globalization of commodity markets, including timber, changes market dynamics that were once driven by local supply and demand. As a result, market forces in one part of the world can lead to forest degradation pressure in far removed regions, including those with large and intact forests. For example, domestic logging bans in China have led to an exponential increase in demand on Southeast Asia's timber producers to supply raw materials for China's rapidly expanding production of processed wood products (Lang and Wan Chan, 2006). The impact of increasing Chinese wood and pulp demand has also been felt in the forests of the Russian Far East. In this region, forest degradation has been particularly rapid as a result of poorly managed logging operations (World Wildlife Fund Forest Programme, 2007). Often, the globalization of timber markets tends to favor large-scale industrial, export-oriented operations. This not only tends to accelerate the rate of forest loss, but it also has a negative impact on the viability and sustainability of smaller operations (Mertz et al., 2005).

Despite rampant forest degradation as a result of unsustainable logging practices, not all timber extraction and international trade in wood products leads to forest degradation or deforestation of these regions. A study of logging throughout the Congo Basin showed that major differences exist between timber concessions based on concession period, size, age, capital source and market focus (Perez et al., 2005a). Large older concessions, particularly those granted to large established foreign entities, tend to utilize formal management plans with a longer term focus. They also tend to harvest trees in a slower, more deliberate fashion than locally financed and local market-focused concessions (Perez et al., 2005). While some of this may be due to governance and land tenure issues, it may also be due to the financial flexibility of concessionaires. Large multinational institutions may have greater flexibility in their capital structure and have greater access to working capital than smaller concessionaires who may be pressured to over-harvest to meet current cash flow needs (Perez et al., 2005a). Forest degradation can also be partially mitigated through the use of reduced impact logging (RIL) techniques, which minimize unnecessary disturbance from harvesting operations. RIL practices have been shown to have fewer carbon losses from logging activities than conventional harvesting operations in the

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tropics, although the benefits of RIL are mostly seen in large scale operations and may not be as significant for small scale harvests (Feldpausch et al., 2005) (see Chapter 9, this volume, for further details on RIL).

### **Foreign investment**

Despite historically low levels of foreign investment in large and intact forests, New Guinea, the Amazon, and the Congo Basin have seen recent increases in foreign investment interests. Foreign investment can be a significant driver of deforestation and degradation if it provides sufficient capital to make certain land conversion projects viable that had previously been uneconomic. Oftentimes, foreign aid packages require economic liberalization policies that lead to increased resource extraction by multinational companies in the particular region. All of the regions discussed in this chapter have received funds from the International Monetary Fund since 1984 (IMF, 2009). In many cases, these loans have provisions for structural adjustment policies, which require the recipient country to allow increased private investment from foreign companies. In Cameroon, structural adjustment policies implemented by international donors following an economic crisis in the 1990s have led to drastically increased foreign investment in the country's natural resources, which has accelerated deforestation and forest degradation (Kaimowitz et al., 1998). In the Congo, financial reforms pushed by international donors and development agencies have largely taken resource management control away from local entities and given multinational corporations greater control over these industries (Kuditshini, 2008). Structural adjustment policies implemented in Indonesia following the Asian financial crisis also increased rates of deforestation and forest degradation (Dauvergne, 2001).

The ways in which these structural adjustment policies impact rates of deforestation and forest degradation, however, are highly complex and interactive (Kaimowitz et al., 1998). Sometimes the presence of large multinational corporations can increase forest governance as these interests seek to protect their own investments through local regulation and oversight. Many multinational organizations have greater transparency in their operations and have active stakeholders who insist that management follow some degree of sustainable practices. This can have a positive effect on logging operations. Still, generally speaking, increased foreign investment leads to increased incremental demand for wood resources, so while governance may be improved, overall deforestation continues simply due to increases in absolute demand for forest products.

### **CONCLUSIONS AND IMPLICATIONS FOR POLICYMAKERS**

Many of the world's large and intact forests have to date avoided significant anthropogenic disturbance due to a number of common factors. These include geographical isolation, low population densities, biophysical constraints on agriculture, a generalized lack of government presence, and low levels of foreign and

domestic investment. It would be a mistake to assume, however, that these forests are not at risk. In the last decade alone, large sections of formerly intact forests on the Indonesian islands of Borneo and Sumatra, on mainland Southeast Asia, and in West Africa have been cleared. Although there is widespread agreement that curbing deforestation and forest degradation in the tropics is critical for many reasons, including the significant carbon releases from these activities, there are few mechanisms to change land management behavior. This in large part is due to the complex mix of deforestation and land degradation drivers. The players, markets, and governance mechanisms are both local and global. Incentives therefore must accommodate the needs of local households while recognizing the roles played by international corporations, banks, and national governments. They must recognize that forest products are a function of both local and global supply and demand forces. As a result, incentives to curb deforestation must be holistic, flexible, and reflect the myriad conditions at both a local and a multinational level.

Given the significant role played by deforestation and forest degradation in widespread global carbon emissions, and the need to reduce these emissions in the face of global warming, countries must make a joint and comprehensive effort to slow rates of deforestation. A primary focus of this effort should be on the world's remaining large and intact forests, particularly in the tropics. Many forest policymakers point out that the developed countries not only contribute the greatest amount of global greenhouse gas emissions, but they are often the key sources of timber and agricultural demand from these sensitive forests. As a result, it has been suggested that developed nations must help to underwrite incentives to compensate developing countries for the opportunity cost of not deforesting, including using carbon market incentives. Emerging mechanisms, including markets for Reducing Emissions from Deforestation and Degradation (REDD) carbon credits, are seen as one example of market-based financial rewards for forest preservation and sustainable management. Without such incentives it may not be possible to stem the tide of forest loss in these regions.

Our policy recommendations for the protection of large and intact forests:

- While countries with large and intact forests share many common variables, there is widespread disparity between governance structures and local drivers of deforestation. Thus, while international forest policies must share a common goal to help prevent continued forest loss, the policies enacted to implement these goals must reflect local conditions.
- In order to concentrate funds where they are needed most, a deforestation risk index should be established to rank and prioritize the disbursement of REDD/REDD+ funds.
- Insecure land tenure is often a driver of deforestation and forest degradation. Countries receiving REDD/ REDD+ funds should be required to have strong, functioning land tenure laws. These rights must extend not only to individuals and forest concessionaires but also to communally governed land.

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- Resource extraction in large and intact forests does not always lead to widespread degradation or deforestation. Avoided deforestation should not preclude reasonable use, including sustainable forestry, hunting, or the use of non-timber forest products.
- Widespread tropical deforestation often occurs in countries that have some degree of infrastructure, an expanding agricultural sector, and an export-oriented economy. Improving existing governance is as important as establishing oversight in countries where there has been little to no governance.
- Road access to core areas of interior intact forest is likely to increase significantly in the near term. International and regional lending institutions should require an integrated forest management plan to limit deforestation and degradation along proposed and existing road networks.
- Management plans should be tailored to the physical, social, and economic realities of the site and should be shaped by the requirements of the funding agency, local governments, and civil society.
- Global mechanisms should be implemented to ensure that forest products in international trade come from sustainable management practices. This may include the use of certification schemes or other designations that identify wood from responsibly managed land.

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## Chapter 16

# Economic Drivers of Land Use Change in the United States

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### EXECUTIVE SUMMARY

While forests in the U.S. have been both a net source and sink for CO<sub>2</sub> at different times throughout history, today they are a weak net carbon sink, largely as a result of changes in land use patterns over time. The capacity of forests to continue to serve as a carbon sink makes them potentially valuable as mitigation tools to offset the damaging effects of greenhouse gas emissions. However, policymakers must recognize that urbanization and development in the U.S. will continually pressure forests, leading to reduced forest cover and fragmented landscapes. From a purely economic standpoint, development is often the highest and best use of land, particularly if financial returns are the primary driver for land use decision-making. Finding the right balance between competing land uses has become an area of focus for economists and policymakers. As policymakers promote carbon strategies for U.S. forests, they should keep in mind what is generally accepted in terms of the economic drivers of land use, and what is less well understood, as outlined below.

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### What is known about the economic drivers of land use change in the U.S.

- Land use change can have a significant impact on carbon storage. While we have seen little net loss of U.S. forestlands in recent decades, increasing pressure to convert forests to other uses has caused concern over decreasing potential for land-based carbon storage.
- Residential and commercial development often represent the “highest and best use” for a parcel of land, resulting in the permanent conversion of forestlands, with negative results on U.S. carbon stocks.

- Subsidies and other government programs alter the balance between forestry, agriculture, and development, including which land use is most profitable at any point in time. Adding forest carbon into the mix of values a landowner can derive from the land may make forests more economically viable.

#### **What we do not know about the economic drivers of land use change in the U.S.**

- The economic viability of forest carbon projects is still unproven. While models have been developed to predict landowner behaviors when carbon is introduced at various prices, these models have not been widely tested. Additionally, price and project risks continue to challenge the economic attractiveness of potential carbon projects.
- Information on land use changes across the country is incomplete. While general trends in land use change can be determined from satellite and remote sensing data, local data at a scale useful to land use planning is not consistently available. Analysis must include not only site specific data, but also local rules and regulatory structures that impact land use behavior.

#### **Some factors to consider when formulating carbon policies for forestry projects:**

- Land use change is the primary driver of carbon sequestration potential in the U.S.
- Residential and commercial development often represent the “highest and best use” for a parcel of land, resulting in the permanent conversion of forestlands, with negative results on U.S. carbon stocks.
- Subsidies and other government programs alter the balance between forestry, agriculture, and development in terms of which is most profitable at any point in time. Adding forest carbon as a way to realize investment returns may make forests more economically viable.
- Whether financially attractive forest carbon projects can be successfully executed is still unproven. Price and project risks continue to challenge the economic attractiveness of potential carbon projects.
- Information on land use changes across the country is incomplete.

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Clearing forestland for development is often a superior economic choice due to higher financial returns versus keeping lands forested.

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## **INTRODUCTION**

In the United States, land conversion from forests to other uses not only has a significant impact on the amount of carbon stored on the landscape, but it often leads to large carbon emissions during the conversion process. Yet, clearing forestland for development is often a superior economic choice due to higher financial returns versus keeping lands forested. In order to create effective carbon policies that help forests compete with other land uses while maintaining them as a large carbon sink, it is necessary to understand the forces driving landowner behavior and choices. This

chapter explores the economic drivers of forestland conversion in the U.S. and the role carbon policy can play in helping forests become a more economically viable land use. It will examine the primary factors contributing to forestland conversion, including shifts in forest ownership and pressure to convert forests to other uses. It will consider the role of carbon-related market incentives and the degree to which they can serve as economic drivers of land use. It will conclude with several policy recommendations for how to improve the economic viability of forests versus conversion to other land uses.

Today, forests cover 33% of the U.S. land base and are an important carbon sink within the nation's total carbon budget. According to the EPA's 2009 greenhouse gas survey, the United States currently emits 6,103 Tg of carbon dioxide (CO<sub>2</sub>), of which 94% is from fossil fuel emissions, primarily electricity generation (USEPA, 2009). Forests, including vegetation, soils, and harvested wood, are currently the largest carbon sink in the United States. Sequestering over 910 Tg of CO<sub>2</sub> (2007), they play a key role in offsetting emissions from other sectors.

While the rate of CO<sub>2</sub> sequestration in U.S. forests has grown nearly 6% annually since 1990, this increase has not been driven by increases in forested area. In fact, according to Forest and Inventory Analysis (FIA) data, the extent of forest cover in the U.S. has not changed significantly since 1900. Rather, forests in the U.S. have matured, resulting in an increase in biomass (or carbon density) on forestland (Woodall and Miles, 2008). This is largely due to forest re-growth from past changes in land use, which are discussed in more detail below.

### **History of forest cover in the United States**

Both the extent of forestland in the U.S. and the rate of carbon sequestration in forests have varied over time, reflecting centuries of changing land use by native populations and European settlers. As human values and resource needs have shifted, forest extent and growth patterns have changed.

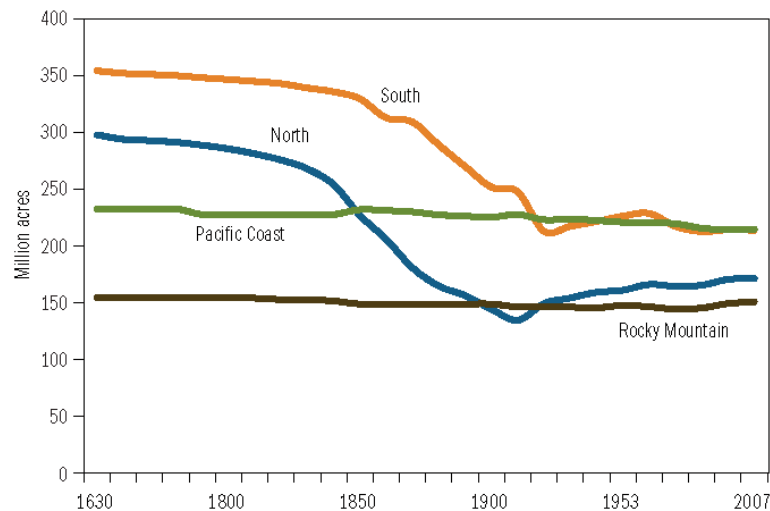
When Europeans began to settle in North America during the 1600s, forests covered approximately one million acres of what is today the United States (Clawson, 1979). Although forest loss during the 17<sup>th</sup> and 18<sup>th</sup> centuries was fairly modest on a national basis, much of it was concentrated in the northeast and to a lesser extent in the southeast where early settlements expanded (Figure 1).

By the early 1800s, however, the trend began to shift. As infrastructure building across the country led to an unprecedented demand for wood products, forests were cut for fuelwood and sawtimber, and more land was cleared for agriculture. This resulted in significant carbon emissions, which continued to grow until they peaked at 2,931 Tg CO<sub>2</sub> annually (not including soil emissions) just after the start of the 20<sup>th</sup> century (Birdsey et al., 2006). Some of these emissions were offset by sequestration in long-lived wood products, nevertheless, this dramatic change in forest cover had a significant impact on the U.S. carbon budget that is still apparent today.

The 20<sup>th</sup> century saw a reversal of this trend as forests shifted from a net source to a net sink of carbon. This occurred largely because landscapes which were once heavily deforested began to regenerate back into forests (Smith et al, 2009).

Regionally, however, the patterns of reforestation following land clearing have been quite diverse. In the South, a large percentage of former pasture and agricultural land has been converted into pine plantations (Sohngen and Brown, 2006). The net impact of this transition (from pre-agricultural clearing to post-agricultural replanting) on forest carbon storage has generally been negative. Due to intensive management practices and, in some cases, the use of genetically altered seedlings, many of these plantations have higher rates of net primary production than naturally regenerating stands (Hicke et al., 2002). However, the intensive management and harvest schedules of these plantations not only cause carbon to be emitted during the harvest process, but the standing forests tend to have lower biomass (and therefore less carbon per hectare) versus natural stands, even when accounting for a portion of harvested materials remaining sequestered in forest products (Sohngen and Brown, 2006). To some extent this has been partially offset by forest encroachment on savannas, also in the southeast. On these sites, carbon storage rates are higher than historical levels (Rhemtulla et al., 2009).

**Figure 1** Changes in forest area by region (1630-2007)



Source: Smith, W., Miles, P., Perry, C., Pugh, S. 2009. Forest Resources of the United States, 2007. US Department of Agriculture, Forest Service, Washington, DC.

In the northeast, land that was originally cleared for timber and agriculture during early colonial history has naturally regenerated back into forest as settlers abandoned farms for more fertile land in the midwest. Carbon stocks on these lands have increased due to forest re-growth, and the forests continue to sequester incremental carbon as they mature and transition to hardwood-dominated stands.

In the western U.S., many of the old growth forests were heavily harvested, but are now federally protected, which has led to significant forest regeneration. Unlike in the northeast, however, carbon in the western forests is primarily concentrated in softwoods



(Hicke et al., 2007). Because of the large extent of publicly owned forestland in the west, in contrast to the northeast and southeast, forest cover and carbon losses in this region today tend to be driven by natural disturbance such as fire and insect outbreaks. Longer term, carbon uptake rates for all U.S. forests will be a function of multiple factors, including soil fertility, stand age, natural and anthropogenic disturbance patterns, and longer term climate effects, including CO<sub>2</sub> fertilization, increased temperature, drought stress, and disturbances such as fire and insect outbreaks,

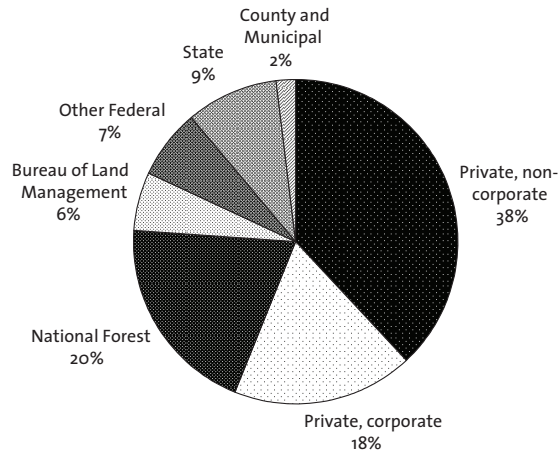
### **ECONOMIC DRIVERS OF LAND USE IN THE U.S.**

The drivers of land use change typically correlate with the highest economic benefits that can be derived from the land, although there are significant variations in this trend across regions of the U.S. and over different historical periods. Forestland values are driven by multiple, interdependent factors. These include timber-related values such as current growing stock, timber prices, local harvest costs, site productivity, and proximity to wood markets, as well as other natural resource values such as mineral and water rights, existing encumbrances, and proximity to development (future demand for real estate conversion). The underlying value of the land (excluding timber related considerations) is a function of population and income growth, housing prices, regulatory and zoning rules and oversight, presence of conservation-focused stakeholders, amenity values, government incentives, and social desires (Murray et al., 2001; Ahn et al., 2002).

The concept of land “rent,” whereby a landowner allocates parcels of land to the use providing the highest level of return, has been used to justify landowner behavior since the theory was first proposed in the 19<sup>th</sup> century by Ricardo and von Thunen. Empirical studies have shown that land rents are the key determinant of most private land use decisions (Ahn et al., 2002). Today, the real estate industry in the U.S. uses the term “highest and best use” or HBU when describing the greatest value that can be derived from a property. Highest and best use is defined as that land use that is legally permissible, physically possible, financially feasible, and optimally productive. For many landowners, selling land to traditional real estate developers often represents its highest and best use. Indeed, corporate owners of timberland often separate out their HBU land from other land managed for timber in their financial reporting, or set up real estate subsidiaries, since land best suited for real estate development carries a higher value than timberland (Weyerhaeuser, 2007).

### **Forest ownership**

Over the last decade, there has been a significant change in the composition of forest ownership in the U.S. This has not only driven changes in the way forests have been managed, but it has altered the landscape itself and, potentially, its capacity for future carbon sequestration. Of the 751 million forested acres in the U.S. today, 44% are in public lands, primarily in the West (Figure 2). The remaining 56% is privately held by families, corporations, conservation trusts, tribes, and financial investors.

**Figure 2 Forestland owners in the United States, 2007**

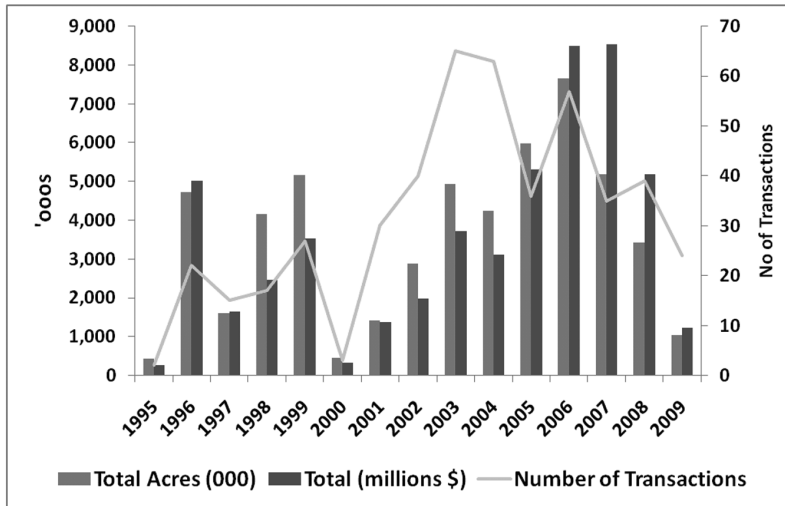
Source: Data from Smith et al. (2009)

Each of these types of forest owners has different, and often complex, reasons for maintaining land in a forested state. The likelihood of their converting their forestland to other uses, such as residential or commercial development, is dependent on diverse factors closely related to their reasons for owning the land, as well as economic and social factors.

Perhaps the most significant shift in ownership patterns in recent years has been large scale land sales by vertically integrated forest products companies to financial investors, including timber investment management organizations (TIMOs), real estate investment trusts (REITs), pension funds, and endowments. The shift began in the mid 1970s when Congress passed the 1974 Employee Retirement Income Security Act (ERISA). ERISA was enacted to encourage pension plans to diversify their investment portfolios away from significant fixed income allocations and into other asset classes, which over time have increased in scope to include real estate and timber (JP Morgan, 2009).

During the 1990s, paper and forest products companies were increasingly pressured to divest their forestland by shareholders who began to recognize the embedded real estate values in their landholdings and demanded that companies sell lands to enhance shareholder returns. Historically, most forest products companies were vertically integrated. Timberland ownership was viewed as a way to secure guaranteed access to raw materials that could be processed at company-owned mills and converted into consumer products. However, as increased real estate demand drove up land prices, and as a growing global fiber supply market offered raw materials at cheaper prices than internal supply, forest products companies began to reconsider the value of holding their timberlands. As pressure mounted from shareholders seeking avenues to unlock the value of land holdings, forest products companies began to divest timberland holdings and/or convert to new corporate structures (Figure 3; Binkley, 2007).

Figure 3 Transactions in U.S. forestland 1995 - 2009



Source: Data from [www.RISI.com](http://www.RISI.com)

While companies such as Kimberly-Clark, Georgia Pacific, International Paper, and Temple Inland divested their timberland, other companies such as Plum Creek, Rayonier, and Potlatch divested their forestry operations and converted into real estate investment trusts (REITs), which are a tax advantaged corporate structure offering high yields to shareholders. Not only are REITs required to pay out nearly all of their net profits as dividends (to qualify for REIT status), but these dividends are taxed at much lower capital gains rates than traditional dividends, which are subject to ordinary income taxes (Chun et al., 2005). This makes them a very attractive investment vehicle for investors looking for regular income. Today Weyerhaeuser is the only remaining large corporate paper and forest products company with large landholdings (Allison, 2006).

Due to favorable regulation that stimulated increasing investor interest in timberland, coupled with corporations eager to sell their landholdings, timber investment management organizations (TIMOs) began to emerge to capitalize on this growing investment asset class and to offer professional management services to investors seeking to implement timber strategies in their portfolios. Over the past few decades, traditional timber companies have sold an estimated 25 million acres of timberlands, largely to institutional investors (Stein, 2005). Today, financial investors, including TIMOs, timber real estate investment trusts (timber REITs), pension funds, endowments, insurance companies, and investment management firms own approximately 40 million acres throughout the country.<sup>1</sup>

The transition from corporate ownership of timberlands for operational needs to financial ownership of timberlands for portfolio returns is likely to have a significant long impact on the way forestlands are bought, sold, and managed. In theory, ownership of timberlands by forest products companies represents a long term, perpetual interest in the land, since its role in the corporation is to supply necessary

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<sup>1</sup> Estimate based on REIT company websites, Yale Program on Private Forests Fact Sheet, "Institutional Timberland Investment," and research conducted by the Open Space Institute.

raw materials for ongoing business operations. On the other hand, the underlying goal of a financial investor is investment gain over a reasonable time horizon, which for most investment funds is typically ten years. Although a significant percentage of forest acreage transferred over the last 15 years still remains as working forest, there is a risk that financial owners will ultimately convert the lands for HBU values, primarily development, to meet high investor return requirements and as an exit strategy when the terms of the investment funds end and capital must be returned to the investors (Rinehart, 1985; Stein, 2006).

While there is already evidence of forestland conversion by institutional investors, many TIMOs explicitly state a commitment to responsible forest stewardship. In fact, some claim that combining timber returns with conservation strategies can be as profitable as more intensive management. This is one reason why certain TIMOs seek to sell working forest conservation easements on their properties. While working forest conservation easements preclude intensive management activities in favor of more sustainable harvest plans, and thus reduce long term timber returns, the sale of an easement to a conservation organization can result in significant cash flow. If this sale is executed early in the life of the property investment, the cash flow will return to investors more quickly, which, due to the time value of money, may yield attractive net returns over the life of the investment. (Stein, 2006; Binkley et al., 2006).

The other significant trend to emerge in forestland ownership is the increasing number of family forest owners, coupled with decreases in average property sizes. Increasing affluence and a desire to own land in rural areas has stimulated sales of forestland to individuals for personal use. Many of these landowners own forested parcels for nonmarket values such as recreation, aesthetics, and a commitment to conservation (Butler and Leatherberry, 2004), confounding traditional economic models of land use decisions, such as the “land rent” concept. Few of these owners actively manage their property for timber or implement long term management plans for their forests. As forest ownership transitions to the next generation, and as more individuals seek landownership in remote forested areas, this is expected to cause increasing forest fragmentation, which in the longer term may impact the ecological resiliency of these lands as well as their carbon sequestration potential.

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### **Forest conversion to development**

Population and income growth put pressure on natural resources and alter the way humans impact their landscapes. This pressure leads to conversion of forestlands to increasingly intensive uses, including housing, commercial development, and natural resource extraction, including energy and food production (Kline and Alig, 2005).

The U.S. Census projects a 35% population increase from 2000 to 2025, including a 79% increase in developed areas, predominantly coming from forests, which are projected to decline by 26 million acres, or 3.5% of total forest area between 2000 and 2030. It is expected that this trend will be most prevalent in the southeast and Pacific northwest, where population growth projections are highest. In contrast, the Rocky Mountains and the Corn Belt are most likely to see a decrease in forest cover as a result of expansion of agriculture, pastureland, and rangeland (Alig et al., 2004).

The demand for land is highest near infrastructure, public and private services, and in places that are relatively affordable to develop in terms of cost and regulatory hurdles. Because agriculture and forestry uses are typically not as financially attractive as traditional real estate development, owners of forest and agricultural land will often sell when there is strong development demand (Kline et al., 2004; Zhang et al., 2005; Mundell et al., 2009). As rural lands are developed, and as infrastructure and impervious cover increase, natural ecosystem functions are compromised. For this reason, federal, state and local governments have established regulations to manage certain types of land conversion and preserve the social and ecological benefits of open space.

Along with population, U.S. household incomes have been increasing. Median household income climbed from \$32,264 in 1994 to \$50,233 in 2008, which has exceeded inflation rates during the period (U.S. Census, 2009). Although populations have grown at the fastest rate in the southeast and west, the greatest increases in incomes have occurred in the southeast and midwest. Second home development, facilitated by higher personal incomes, has been a driver of forest fragmentation in areas close to major metropolitan centers, such as the Catskills in New York (Tyrrell et al., 2005).

### Change in forest area with development

Although net forestland area has remained fairly constant over the past several decades, almost 50 million acres were converted out of and back into forest cover between 1982 and 1997 (Alig et al., 2004). Consequently, some forests have undergone dramatic change. Lands that were deforested were largely mature forests with significant carbon pools and structural diversity. The forests regenerating in their place today are much younger, have lower biomass values, and, if regenerated as plantations, also have reduced structural diversity<sup>2</sup> (Alig and Plantinga, 2004; Wimberly and Ohmann, 2004).

### Urban and rural sprawl

*Urban sprawl* (sometimes called “suburban sprawl”) is the term used to describe the conversion of open lands to development in a sprawling pattern, typically radiating out from a metropolitan area. The first major trend in suburban sprawl took place following World War II (Radeloff et al., 2005). Greater personal use of automobiles and road funding from the Federal Aid Highway Act of 1956 increased access to the fringes of metropolitan areas, which led to greater low-density development (Jeffords et al., 1999). *Rural sprawl*, on the other hand, refers to scattered houses and other structures built on landscapes in non-metropolitan areas. Typically, rural sprawl occurs in areas with high natural amenity values. Since the 1970s, this type of growth has been driven by a desire by homeowners to purchase primary or vacation homes in naturally beautiful areas with affordable transportation connections to metropolitan areas (Radeloff et al., 2005; Ward et al., 2005). These two development trends differ, in that urban sprawl impacts less total area than rural sprawl but has

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Because agriculture and forestry uses are typically not as financially attractive as traditional real estate development, owners of forest and agricultural land will often sell when there is strong development demand.

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<sup>2</sup> The dynamics of carbon sequestration as these stands develop is discussed in more detail in Chapter 3 in this volume.

more intense effects, whereas rural sprawl can have a much larger spatial impact, though it may be less intense on a particular parcel of land (Radeloff et al., 2005). Carbon emissions from sprawling development occur immediately as forests are cut down for houses, and at a higher rate over a longer period of time as a result of increased fossil fuel burning from personal vehicles traveling greater distances.

Lands that are close to dense population centers, and are in proximity to affluent areas, natural amenities, and public services such as water and electricity, are more susceptible to conversion (Plantinga et al., 2001; Ahn et al., 2002). The financial incentive to sell forestland in these areas can be very strong. For example, in the southeast and the Pacific Northwest, forestland is worth 25 to 141 times less (respectively) than urban land values (Alig et al., 2004). As the relative profitability of selling rural land for development grows versus retaining land in a forested or agricultural state, landowners often accelerate sales to capture high valuations (Alig, 2007).

### **GOVERNMENT POLICIES AFFECTING RATE OF FOREST CONVERSION TO DEVELOPMENT**

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Although net forestland area has remained fairly constant over the past several decades, almost 50 million acres were converted out of and back into forest cover between 1982 and 1997

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Government policies can both mitigate and exacerbate development pressure on rural lands. Federally owned forests are currently well protected from development pressures. Typically, they are managed for a variety of uses, including timber extraction, recreation, watershed health, and wildlife habitat. Private forests are much more susceptible to conversion. Development rates on these lands are impacted by federal, state and local policies, including forest protection incentives, tax incentives, zoning, transportation funding, infrastructure projects, and mortgage incentives, among others.

Federal forestry assistance is provided through United States Department of Agriculture (USDA), either through the Farm Bill or through the United States Forest Service (USFS). As of 2008, assistance included technical and financial aid for forest management, forest protection, forest recovery and restoration, and economic assistance. The Forest Legacy program specifically authorizes the Forest Service to protect forestlands at risk of conversion to development or agriculture by purchasing the lands or funding a conservation easement on the property. Similarly, the Community Forest and Open Space Conservation program, established by the 2008 Farm Bill, provides funding to purchase titles to at-risk forestlands. State funding administered through state environmental agencies provides complementary funding for private landowners to keep their land forested and help support proper management of these properties. Additionally, various tax incentives are targeted at non-industrial private forest owners to reduce the cost burden of forest management activities (Riitters et al., 2002; Gorte, 2007).

While these initiatives have helped to protect forestland and promote healthy forested landscapes, other policies have facilitated conversion of forestland to urban and rural sprawl-related uses, sometimes unintentionally. Policies designed to stimulate economic development may have unintended negative consequences for forests if they cause increased urban and rural sprawl. The most commonly cited

federal drivers of sprawl include highway spending, water and sewer system requirements, and subsidies for suburban homeownership. Other government policies can contribute to sprawl as well, including economic development incentives such as income tax credits or local zoning regulations that lack provisions to protect open space. Overall, it would appear that there is a tension between government programs to combat sprawl, and subsidy and regulatory programs which encourage sprawl, either intentionally or unintentionally (Jeffords et al., 1999). Still, when the U.S. General Accounting Office (now the Government Accountability Office) reviewed the influence of these federal policies on sprawl, they concluded that their impact is “unclear” (Jeffords et al., 1999). In the end, job creation, affordability and public and private services are often a greater priority for governments than managing sprawl.

### Fragmentation

Rural sprawl has a unique impact on forests by physically fragmenting, or breaking up, large contiguous patches of forestland with houses, roads, and other development. Breaking forests into smaller patches creates more forest edges and fewer forest interiors, even if the developed area among the patches is relatively small (a country road, for example). Currently 62% of forestland in the contiguous United States is located within 150 meters of a forest edge (Riitters et al., 2002).

Increasing forest edge and reducing interior area can compromise ecological values, which may in turn influence land values. In fragmented forests, certain plants, animals and insects gain a competitive advantage from increased light or a change in nutrient cycling. For example, invasive plants and animals which can thrive along forest edges may out compete native species, particularly if invasive plants are better able to weather roadside runoff laden with salts and other pollutants. Increased access to forests can lead to illicit dumping and campfires, which can create hazards for neighboring communities and increase the property's susceptibility to wildfire (Theobald and Romme, 2007). As a landscape becomes fragmented, its resilience to large disturbances may be reduced. This may in turn threaten developments at the edge of forested landscapes. Additionally, ecosystem services such as water quality or stormwater catchment may be reduced by fragmentation. In these ways, fragmenting landscapes may have consequences that can ultimately impact the value of developed lands. Unfortunately, it may be difficult to clearly link the long term effects of fragmentation to ecosystem services values. In addition, preventative measures require long term land management planning, which may be outside the time horizon of most homeowners and local zoning officials.

At the same time, residential development near working forests can pose difficulties for forestry operations. New homeowners who move close to the forest for its amenity values, such as scenic vistas and tranquility, may not want to see and hear forest operations in the area. Frustrated with the noise and pollution associated with forestry activities, residential landowners may require forest managers to alter their management plans either by altering transportation routes for harvested wood, dictating when harvesting activities can take place, and precluding certain

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Policies designed to stimulate economic development may have unintended negative consequences for forests if they cause increased urban and rural sprawl. The most commonly cited federal drivers of sprawl include highway spending, water and sewer system requirements, and subsidies for suburban homeownership.

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management strategies such as prescribed burning. Neighbors may even file expensive nuisance lawsuits against forest owners in order to restrict forestry activities (Alig and Plantinga, 2004). This can impact both the productivity of the landscape as well as the cost structure for the forest manager. This may make forest management a less profitable activity in these areas, further reducing the economic viability of working forests and increasing the pressure to convert to other land uses.

### **Parcelization**

Parcelization is the division of a large property into smaller land holdings more appropriately sized for development. Landowners are often driven to parcelize their land to increase its market value because land can be sold at higher prices if it is subdivided, since the embedded value of the individual parcels is generally reflected in the sale price. If the purchaser retains the land as one parcel, the carrying cost of the property will be higher since the buyer has paid for the parcelization and continues to pay taxes that reflect the higher value. This may not only pressure the buyer's desire to sell some of the parcels to reduce the carrying cost, but it may make it unprofitable to engage in any ongoing forestry activities if cash flows from timber sales cannot match the higher carrying costs of the land. Thus, although parcelization allows landowners to monetize the development value on their land without actually developing the land, the result is that forestry and agricultural uses may be priced out of the land use options even if the property remains intact (Alig and Plantinga, 2004).

If a purchaser of the land sells one or more parcels for cash flow and manages the rest of the property for forestry, they will likely face a different profit structure on smaller acreage than they would have on the larger parcel. Larger parcels create economies of scale that can help to spread the fixed costs of equipment and administration over a larger land base. Moreover, the larger the parcel, the more flexibility a landowner may have in the timing and scale of harvests and in generating cash flows from multiple uses. Insufficient scale may also make it difficult to implement viable projects for carbon sequestration, water quality, and wildlife habitat. (Plantinga et al., 2001).

### **Managing sprawl through land conservation**

Concern over the lack of open space and the loss of the ecological and social values it provides has increased as more forestland is fragmented or lost to development. The amount of forestland per capita in the U.S. has declined by almost half in the past 50 years, and by as much as two thirds in the Pacific northwest, largely due to population growth (Kline et al., 2004). In fact, in April, 2007, forestland per capital fell below 1 ha for the first time in U.S. history (Woodall and Miles, 2008). Historically, government policies to protect public lands through national and state forests and parks have been instrumental in protecting open space for the public good (Alig, 2007). However, these policies have not helped to protect private lands. Since real estate markets have not been able to efficiently value the social benefits of private forest land, including aesthetics, recreation, water resources, and resilient ecosystems, they are often left out

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of land buyer decision-making processes. (Kline et al., 2004). For this reason, there have emerged policies specifically designed to encourage open space preservation, such as urban growth boundaries, open space set asides, and tax incentives for keeping forests intact (Plantinga et al., 2001).

However, such policies can have unintended consequences. In their 2003 study on the influence of public space on urban landscapes, Wu and Plantinga (2003) found two potential sprawl effects from designating public open space outside the city boundary. First, they showed that urban residential communities may expand to envelope the open space. This is particularly likely if the open space is sufficiently close to the city (ease of access) and if it provides a high level of amenities. A second possibility is that open space set asides create "leap frog development," bypassing the restricted open space to build further away from existing centers of development (Wu and Plantinga, 2003). Ironically, their research suggests that while delineating public open space may protect some portion of land, it may inadvertently encourage outward sprawl that is greater than what would have occurred without the set asides.

### **GOVERNMENT POLICIES AFFECTING FORESTRY AND AGRICULTURAL LAND USES**

Markets for both forest and agricultural land are influenced by government policies, including agricultural subsidies, conservation programs, and timber harvesting regulations. These policies exist to promote economic livelihoods, but also to protect socially desirable quantities of certain land types and ecosystem processes. They can also alter land use decisions by influencing commodity prices and the supply of and demand for natural resources (Alig et al., 1998; Ahn et al., 2002). For example, research suggests that in some cases agricultural subsidies and guaranteed prices paid to farmers have promoted more forest conversion to agricultural land than would have taken place without such subsidies (Alig, 2007). The impacts of government programs and policies on land use change are enormously complex and vary by program.

#### **The U.S. Farm Bill**

Competition between forestry, agriculture, and other land uses has been heavily driven by government subsidies since the New Deal in the 1930s. At that time, the U.S. government intervened in agricultural markets by providing subsidies to farmers to stabilize crop prices. The Agricultural Adjustment Act (AAA) was passed in 1933 to raise prices for basic agricultural commodities in order to increase the rent attained on agricultural lands (Skocpol and Finegold, 1982). The AAA was the first of many federal agricultural subsidy programs. The U.S. Farm Bill has since replaced the AAA as the government's predominant food and agriculture policy tool. The practice of subsidizing agricultural products has increased over time; in fact, between 1997 and 2006, government payments accounted for 30% of net farm income (Jordan et al., 2007).

Farm subsidies that favor agriculture inflate the agricultural land value relative to other uses. While agricultural subsidies may reduce the likelihood that agricultural

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Since real estate markets have not been able to efficiently value the social benefits of private forest land, including aesthetics, recreation, water resources, and resilient ecosystems, they are often left out of land buyer decision-making processes.

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land is converted to developed land uses, it may increase the likelihood that other rural lands, such as forests, will be developed. While the Farm Bill continues to support agricultural subsidies, recently, new provisions have been added to address the importance of maintaining working forests. For example, the 2002 Farm Bill authorized the Forest Land Enhancement Program, a multi-million dollar forestry program designed to assist non-industrial private forest landowners. The current 2008 Farm Bill (Food, Conservation, and Energy Act of 2008, P.L. 110-246) has taken a step further and outlined a set of principles specifically aimed at forest stewardship. These include:

- Conserving and managing working forests for multiple values and uses;
- Protecting forests from threats, including “catastrophic wildfires, hurricanes, tornados, windstorms, snow or ice storms, flooding, drought, invasive species, insect or disease outbreak, or development,” and restoring appropriate forest types in response to such threats;
- Enhancing public benefits from private forests, including air and water quality, soil conservation, biological diversity, carbon storage, forest products, forestry jobs, production of renewable energy, wildlife, wildlife corridors and wildlife habitat, and recreation.

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While agricultural subsidies may reduce the likelihood that agricultural land is converted to developed land uses, it may increase the likelihood that other rural lands, such as forests, will be developed.

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### **The Conservation Reserve Program**

The U.S. Department of Agriculture’s Conservation Reserve Program (CRP) subsidizes the maintenance of ecosystem functions by compensating farmers for converting erodible and ecologically sensitive agricultural land into alternative land uses such as forest or grassland (Roberts and Lubowski, 2007). Established in 1985, the program has preserved approximately 34 million acres for up to 15 years (Sullivan et al., 2004), of which approximately 7% was put into forest use (Plantinga et al., 2001).

The CRP is often held up as a model for how the government might create incentives for other ecological values, including carbon sequestration. Some researchers claim that from 1986-1991 the CRP effectively made land conservation a competitive choice from a market perspective (Plantinga et al., 2001). During this time, CRP administrators specified maximum allowable rental rates for retiring agricultural lands by region while allowing unlimited enrollment of lands. By using farmers’ behavioral responses to the CRP as a proxy for likely actions under a carbon market, the authors (Plantinga et al., 2001) suggest that carbon management will require an annual return threshold of \$40-\$80 per acre (depending on the region) in order to stimulate conversion of agricultural lands into reforestation projects.

### **Timber management regulations**

Pressure to protect forested areas for wildlife and other ecological services has led to government policies that restrict timber extraction on public lands. While this has ensured protection of habitat on public lands, it has often led to timber harvesting activities being diverted to unregulated private lands. For example, in the Pacific

northwest, protection of the endangered northern spotted owl shifted timber harvests to other parts of the country, particularly the southeast. At the same time, reduced harvesting on public lands has led to significant build up of fuel loads in federal forests, hence increasing the risk of wildland fire and its associated greenhouse gas emissions. It has also led to more dense forests, which are at greater risk of disease and pest outbreak. Curtailments on harvesting on federal lands have simply shifted harvesting and its commensurate carbon emissions elsewhere while increasing the risk that existing carbon stocks on public land may be lost to large scale disturbance.

Recently, policies have been proposed to regulate timber harvests on private lands to protect ecosystem values. Research has shown that extended timber rotations provide greater stand structure and age class diversity, which is positive for wildlife and carbon management. Proponents of policies designed to lengthen timber rotations claim that, in the long term, forestry land rents should increase, since more mature wood commands a higher price in timber markets. This may create an effective incentive for landowners to keep lands in a forested state for a longer term (Alig et al., 1998).

### **Agricultural biofuels subsidies**

Due to greater national interest in renewable energy sources, liquid biofuels (or simply “biofuels”) have become an increasingly attractive land management option. As demand for biofuels has increased, this has put pressure on forestlands to convert to agricultural biofuel production (Murray, 2009). Biofuels (e.g. biodiesel and ethanol) are fuels that are derived from organic matter. In the U.S., biofuels are primarily produced from feedstocks such as corn, soybeans and sorghum. In many countries around the world, including the U.S., use of biofuels has been mandated for transportation fuels. Increased demand for biofuels in recent decades has led to what a recent report from the International Institute for Sustainable Development called “frenzied” expansion in the industry (Koplow, 2006). Because of major subsidies at all levels of the supply chain, many rural landowners have shifted into production of these lucrative crops. In 2007, 24% of corn harvested in the U.S. went to ethanol production (Howarth and Bringezu, 2009). As biofuel production increases, so does demand for agricultural land, which shifts economic rents in favor of fields over forests. While it is unclear today how biofuel subsidies will impact land use decisions longer term, they have raised significant concern among stakeholder groups interested in preservation of forestland and agricultural land for food crops (Koplow, 2006). Ironically, high demand for biofuels may have unintended carbon implications. Not only does land cleared for biofuel production lead to greenhouse gas emissions during the conversion process, but the increased use of nitrogen fertilizers prevalent on land managed for biofuels may also offset the carbon savings from substituting biofuels for fossil fuels (Howarth and Bringezu, 2009).

Still, forestland owners may benefit from biofuels subsidies through incentives to encourage the use of wood biomass. This could serve to increase the economic returns from forestland as new markets emerge for wood fuels, particularly for thermal use in residential, commercial, and public structures. However, while

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increased wood fuel use may reduce carbon emissions from the energy industry, it may also compromise carbon sequestration rates on standing forests, particularly if forests are transitioned to shorter rotation plantations to serve the wood fuel market.

### Carbon storage

Over the past decade, a new potential economic driver has emerged: carbon sequestration. While forests were not included in initial agreements to mitigate global climate change, they have since been acknowledged as important carbon sinks by scientists, policymakers, and government officials. There is much interest in pushing for the inclusion of forests in the next iteration of a United Nations Framework Convention on Climate Change (UNFCCC), which is expected to replace the Kyoto Protocol when it expires in 2012. Meanwhile, the U.S. Congress continues to debate new federal climate change legislation, including the role the forests might play in future mitigation efforts.<sup>3</sup>

It is much too early to know the impact that carbon related policies will have on forests and forestland conversion in the U.S. Among carbon market architects and participants, there is a high level of uncertainty over the economic viability of forest carbon as both a climate mitigation tool and as an economic incentive to preserve forestland. Pricing, project costs, risk management, and the balance between competing land uses all arise as key issues when exploring the potential for forest carbon sequestration as an economic and policy tool.

<sup>3</sup> See Chapter 17, this volume for a discussion of forests and climate change policy in the U.S.

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While it is unclear today how biofuel subsidies will impact land use decisions longer term, they have raised significant concern among stakeholder groups interested in preservation of forestland and agricultural land for food crops.

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### *Pricing and transaction costs*

Pricing trends for carbon credits around the world have been mixed. In recent years, carbon credits have ranged in value from approximately US\$1.50-\$35/ton in the international regulatory market (Capoor and Ambrosi, 2007). Transactions in the U.S. voluntary market have been even more variable, ranging from US\$1.20-\$300/ton (Hamilton et al., 2009). The lack of both price stability and clear rules for market participants have inhibited many potential carbon project developers.

At the same time, transaction costs for creating carbon offsets have remained high. According to economists van Kooten and Sohngen (2007), costs associated with forest carbon projects have ranged from \$3-\$280 per ton of carbon. Important factors contributing to this disparity are the location of the project, the project type (reforestation, avoided deforestation or managed forest), whether opportunity costs have been included, and the methodology used to measure the carbon itself. Despite the wide variety of costs, and the high probability that they may exceed expected carbon revenues, efforts to promote forest carbon projects remain strong. Proponents often cite current high levels of expertise in forest management (particularly in developed countries) alongside an expectation that new technologies will emerge to lower the overall cost structure of forest carbon projects (Murray, 2009).

Interestingly, carbon payments have been shown to increase private forestland acreage in nearly all models of carbon pricing scenarios. However, optimal acreage for carbon sequestration may not be the optimal mix of agriculture and forests for larger

economic concerns beyond private landowner interests. Climate change mitigation impacts need to be viewed against changing demographics and welfare needs to determine the most appropriate use of lands for multiple human values and needs (Daigneault et al., 2009).

## **CONCLUSIONS**

Economic drivers have influenced land use in the United States since the first European settlers arrived. Carbon stocks have fluctuated as land has been forested, managed for agriculture, or converted to development. Whereas U.S. forests served as carbon sources during the 1800s, changes in land use over the past 100 years have transformed U.S. forests into a net carbon sink. Many landowners, scientists, and policy makers acknowledge the potential for these forests to capture greenhouse gas emissions and are seeking to determine the appropriate mixes of land use for carbon sequestration and other values, considering both economic and ecological factors. Current drivers of land use change include demographics, market demands, government policies, and owner preferences, and others are complex and ever changing. Whether or not carbon-related incentives will alter land use decision-making is still undetermined. While the introduction of carbon markets is often seen as a potential driver to help forests compete with other land uses, the economic viability of forest carbon projects remains undetermined.

### **Management and policy implications**

As the debate continues over the role of forests in climate change mitigation, scientists and policy makers will need to consider several unanswered questions:

- What policies would give forests (as carbon sinks) a competitive economic advantage over other land uses?
- How might using forests to mitigate climate change impact ecosystem services and other valued land use options?
- What are the larger impacts of forest carbon projects on public welfare?
- How will we measure and model land use change? Are there tools that provide simple, accurate, accessible information for policy makers?

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## Chapter 17

# United States Legislative Proposals on Forest Carbon

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### EXECUTIVE SUMMARY

This chapter provides an overview of the role of managing forests to store carbon in the efforts to adopt U.S. climate legislation at the national level (as of June 2009). While the U.S. has not ratified the Kyoto Protocol or adopted national climate legislation yet, considerable efforts have been underway to reduce emissions of greenhouse gasses at the regional (Northeastern U.S.), state (California), municipal, corporate, and individual levels. The issue of storage of carbon in forests and farmland has played a major role in U.S. emission reduction efforts, particularly in the voluntary carbon markets. As the demand for land-based carbon offsets has grown, so too has the demand for rules to define high quality, real offsets. The U.S. market has responded with a range of such rules, from those directly supported by governments, to those that are purely voluntary. Some of these rules cover how best to account for carbon in forest systems, such as: the types of forests/forestry operations covered; the pools of carbon in the forest that are included; the location of acceptable projects; and the “business as usual”/baseline emissions to be considered. Others go more directly to the quality of the offset produced, namely, whether the emission reductions are truly “additional” to those that would have happened anyway; how best to monitor and verify that the promised storage has occurred; how to protect against “leakage,” i.e. that the emissions just move to another location; and how to ensure that the storage is permanent or how to protect against potential releases in the future. As federal efforts to adopt climate legislation intensify, these lessons learned from the voluntary carbon markets are being incorporated into the draft bills. It is clear that any U.S. federal climate legislation will include provisions to encourage the storage of carbon in forest and agricultural lands – both through the markets for carbon offsets, as well as direct public funding. The

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details of these programs, however, are likely to be delegated to the U.S. Department of Agriculture and other federal agencies to be worked out.

### WHAT WE KNOW AND WHAT WE DON'T KNOW

While it is extremely difficult to predict how U.S. federal climate policy will evolve, there are a few areas where the likely results seem clear:

**Inclusion of forests:** If and when the U.S. adopts federal climate policy, forests and other land uses are likely to play a major part in both the market and public funding approaches adopted.

**Scope of forest systems included:** A wide range of land uses seem likely to be included, such as afforestation/reforestation and managed forests, as well as soil carbon in farm and range lands. The inclusion of harvested wood products as approved project activities seems less likely at this time.

**Spatial scale:** Both domestic and international offsets from forest projects seem likely to be included. One open question is whether credits from international projects should be discounted compared to those from domestic projects.

**Quality assurance:** Substantial requirements will be imposed to help ensure that the offsets are “real.” Finding the right balance between lower cost and higher accuracy will be difficult in the areas of monitoring and verification.

**Leakage:** While any policy will refer to the need to address leakage, few concrete measures to do so outside of project or entity boundaries seem likely to be required.

**Permanence:** Some combination of dedicating land to carbon storage for a lengthy period of time (through a conservation easement or contractual arrangement) and requiring that a portion of the credits be held for use as a buffer against unexpected changes seems likely. While there is some discussion of temporary credits, experience in the CDM market suggests that other ways should be used to address permanence issues.

What we do not know about the role of forests in likely future climate policy in the U.S. is a much larger set of questions, encompassing not only the unresolved scientific questions covered in other chapters in this volume, but also the constantly shifting efforts to build political coalitions in favor of federal legislation.

**Keywords:** U.S. climate change legislation, Waxman-Markey bill, Climate Action Reserve, RGGI, voluntary carbon markets, Chicago Climate Exchange, Voluntary Carbon Standard, DOE 1605(b) program, public funding for carbon sequestration

### INTRODUCTION

#### The United States and climate change policy

In the past decade, climate change has moved to the forefront of environmental concern in the United States. While 183 countries have ratified the Kyoto Protocol to the UN Framework Convention on Climate Change (UNFCCC, 2009), the U.S. has

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It is clear that any U.S. federal climate legislation will include provisions to encourage the storage of carbon in forest and agricultural lands – both through the markets for carbon offsets, as well as direct public funding.

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not. Among the reasons given are concerns that the Kyoto Protocol does not set realistic goals and does not include emissions from rapidly growing developing countries (Barrett and Stavins, 2003).

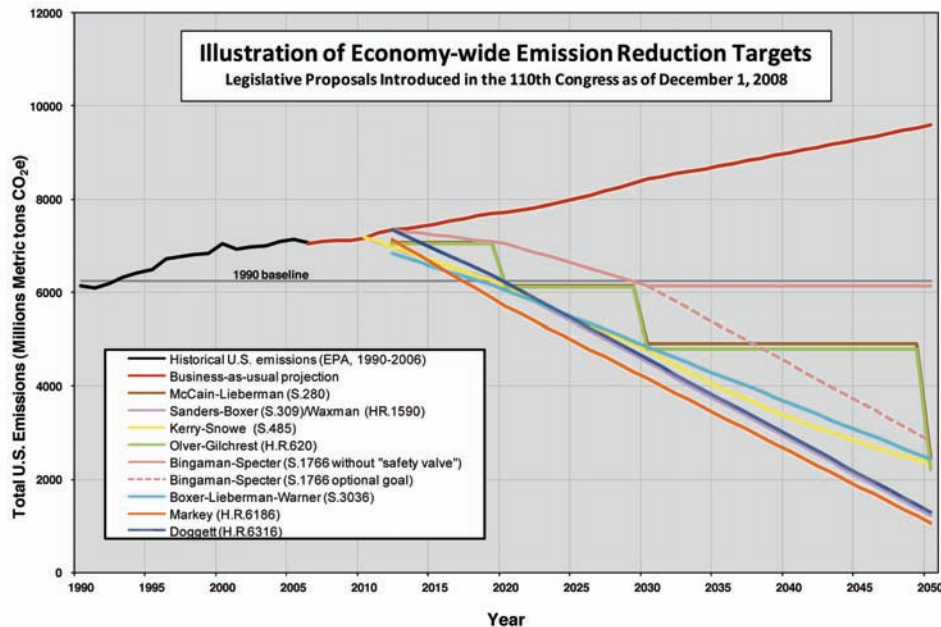
In the absence of action by the U.S. at the international and national levels, regional, state and municipal climate initiatives have emerged, along with voluntary efforts. For example, at the local level, on February 16, 2005, the date the Kyoto Protocol became law in 141 countries, Seattle Mayor Greg Nickels launched the U.S. Mayors Climate Protection Agreement. The agreement represents a local effort to meet or beat Kyoto Protocol targets in communities across the U.S. By 2008, 916 cities and towns from 50 states, Washington D.C. and Puerto Rico had joined the Mayors Climate Agreement, representing more than 83 million citizens (U.S. Conference of Mayors, 2009). At the state level, in September 2006, Governor Schwarzenegger of California signed the Global Warming Solutions Act, making California the first state to cap greenhouse gas (GHG) emissions in the U.S. (California AB32 2006). Similarly, under the Regional Greenhouse Gas Initiative (RGGI), ten Northeastern and Mid-Atlantic states have agreed to cap and reduce emissions from the power sector by 10 percent by 2018 (RGGI, 2009). As of 2009, RGGI is the first mandatory, market-based effort in the United States.

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In the absence of action by the U.S. at the international and national levels, regional, state and municipal climate initiatives have emerged, along with voluntary efforts.

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Figure 1 Future emission reductions considered in the 110<sup>th</sup> Congress



Source: Pew Center on Global Climate Change, 2008. Economy-Wide Cap-and-Trade Proposals in the 110<sup>th</sup> Congress. Reprinted with permission.

Many argue that it is in the best interest of the U.S. to develop a national GHG program that will allow the U.S. to be part of any future global climate agreements, particularly after the Kyoto Protocol expires in 2012. The Obama administration and

the democratically-controlled Congress share this view. As a result, there has been a surge in efforts to design a national emissions cap and trade program, as part of the federal response to climate change. For example, a consortium of major corporations (e.g. Alcoa, BP, DuPont, GE, Pepsi, Shell) and leading environmental groups (e.g. World Resources Institute, Natural Resources Defense Council, Environmental Defense Fund, The Nature Conservancy) formed the United States Climate Action Partnership (USCAP) “to call on the federal government to quickly enact strong national legislation to require significant reductions of greenhouse gas emissions” (USCAP, 2009a). The negotiations behind USCAP’s Blueprint for Legislative Action have influenced the design of many recent Congressional proposals (USCAP, 2009b).

As of December 2008, there were ten economy-wide cap-and-trade proposals under consideration in the 110<sup>th</sup> Congress (Pew, 2008). Figure 1 shows the emission reduction goals of the ten proposals. In the 111<sup>th</sup> Congress, it is expected that the Boxer-Lieberman-Warner Climate Security Act (U.S. Congress, 2008) in the Senate and the Waxman-Markey American Clean Energy and Security Act of 2009 (U.S. Congress, 2009) in the House of Representatives will provide leading examples of the climate proposals around which negotiations will occur.

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In an effort to lend some structure to the voluntary carbon markets, a number of different organizations have developed rules for ensuring that offsets actually lead to emission reductions.

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### **The voluntary carbon market in the United States**

At the same time that the municipal, state, regional and national efforts to address climate change in the U.S. have expanded, so too has the work by corporations, academic institutions, individual U.S. citizens and others to reduce their carbon footprints. One part of these efforts is an active market for voluntary carbon offsets – for example where the owner of a car pays to have a farmer reduce GHG emissions from farm operations (Hamilton et al., 2009). Many corporate buyers in the voluntary market participate in order to better understand the transaction process in anticipation of a federal cap and trade system that includes offsets.

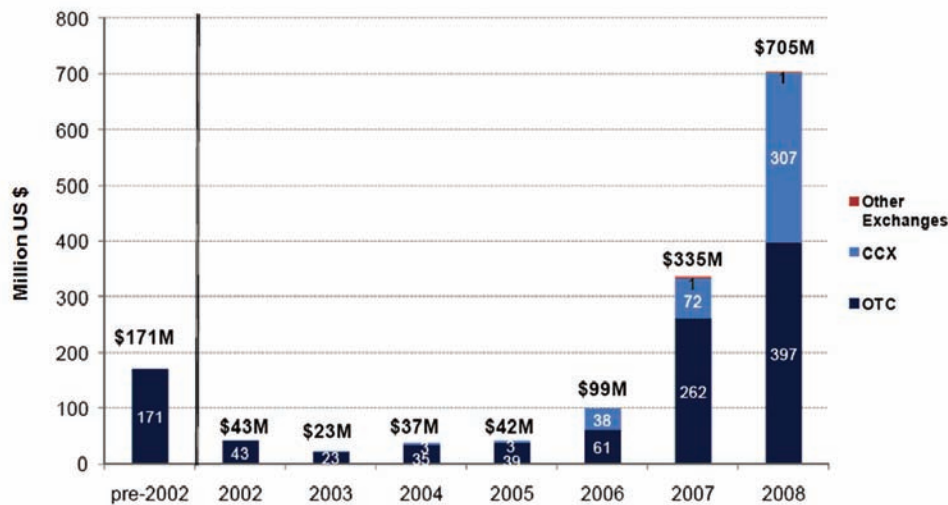
In an effort to lend some structure to the voluntary carbon markets, a number of different organizations have developed rules for ensuring that offsets actually lead to emission reductions. It is expected that the standards and guidelines ultimately included in a federal GHG regime will draw heavily from the experience and rules being used in the voluntary carbon markets.

While the first voluntary carbon transaction in the United States occurred in 1988 (sixteen years before the first one in the Kyoto Protocol), when carbon offsets were purchased from a forestry project (Hamilton et al., 2009), transactions in the over-the-counter (OTC) market did not gain significant momentum until the early 2000s (see Figure 2). The launch of the Chicago Climate Exchange (CCX) in 2003 added further depth to the voluntary carbon markets. In 2005, voluntary markets scaled up as offsetting emissions entered the mainstream and there was an increase in transactions, as well as both praise and criticism.

Ecosystem Marketplace and New Carbon Finance tracked a total of 66 million tons of carbon dioxide equivalent (MtCO<sub>2</sub>e) worth of offsets traded in the U.S. voluntary carbon market in 2007 and 123.4 MtCO<sub>2</sub>e in 2008 (Hamilton et al., 2009). Of the 123.4 MtCO<sub>2</sub>e traded in 2008, 54 MtCO<sub>2</sub>e (44%) involved exchanges on the

OTC market and 69.2 MtCO<sub>2</sub>e (56%) were conducted on the CCX. The total value of the transactions for the year was U.S. \$705 million (Hamilton et al., 2009).

**Figure 2** Historic growth in the voluntary carbon market



Source: Hamilton, K., Sjardin, M., Shapiro, A., Marcello, T., 2009. A Fortifying the Foundation: State of the Voluntary Carbon Markets 2009. A report by Ecosystem Marketplace and New Carbon Finance, May 2009. Reprinted with permission.

### Forests as part of the U.S. climate change strategy

Forests influence greenhouse gas concentrations because they are both a potential CO<sub>2</sub> sink (sequestering carbon) when they grow, as well as a potential source of CO<sub>2</sub> when they are disturbed. According to a 2007 report from the Intergovernmental Panel on Climate Change (IPCC), deforestation and subsequent land use change accounted for 17.4% of total anthropogenic global greenhouse gas emissions in 2004 (IPCC, 2007). Any comprehensive climate change policy must address these emissions.

At the same time, forests have a significant potential to sequester carbon. Compared to alternatives such as industrial carbon capture and storage, forest offset projects are regarded as a less expensive means of carbon storage (Enkvist et al., 2007). Moreover, financial incentives for carbon sequestration in forests would help fund biodiversity conservation efforts in the U.S. and abroad. The “technology” or ability to sequester and store carbon in forests already exists, and keeping forests as forests (i.e. preventing deforestation) is the most straightforward way of maintaining carbon stocks. Given its large land base, the U.S. is also well positioned to use its domestic forests to help meet any future national emission reduction targets. If carbon storage in farm and range lands is included, this may help acquire votes from senators in the key Midwestern states in favor of U.S. climate legislation.

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Given its large land base, the U.S. is well positioned to use its domestic forests to help meet any future national emission reduction targets.

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For these reasons, many of the proposals for U.S. climate change legislation include incentives – either through the carbon markets or public funding – for activities that increase forest carbon sequestration and reduce emissions from deforestation and degradation. If such incentives are approved, they would go beyond the existing structures of the Kyoto Protocol and the Clean Development Mechanism (CDM), which have not yet included forests as a significant source of offset projects (for a discussion of global policies, see Chapter 18 of this volume). While the rules under the Kyoto Protocol include forests as a verifiable GHG sink, tradable credits are only granted for afforestation/reforestation projects established after 1990 (European Commission, 2009). Moreover, these credits under Kyoto are “temporary credits” and typically trade at a discounted price (Hamilton, 2009).

Generally speaking, the prices paid for offsets in the U.S. voluntary carbon markets are lower than those paid in international compliance markets, such as in the EU’s Emissions Trading System (Carbon Positive, 2009). However, forestry credits have remained some of the highest priced offset credits in the U.S. For example, in 2006 and 2007, credits from afforestation/reforestation projects received the highest prices of any offset projects in the U.S. voluntary markets (Hamilton et al., 2007). Moreover, many (though not all) forest carbon projects have a higher value on the OTC market for the social and environmental co-benefits they offer. For companies buying voluntary credits for the sake of public relations, the tangible nature of conserved land and general understanding of trees in the carbon cycle adds to their appeal – so-called “charismatic carbon” (Hamilton et al., 2007; Conte and Kotchen, 2009).

However, even in the U.S., the groups setting rules for and creating registries to track offsets have faced difficulties incorporating forest carbon projects into their frameworks. This is due, in part, to the variable nature of forest growth and the risks associated with the potential impact of natural disasters (e.g. fires or disease) on carbon stocks. Given the complexity and variability in forest systems, forest-based offset projects have raised many debates over how to account for and ensure the quality of the credited emission reductions over time.

In addition, forestry or land use offset projects can involve a number of different types, each posing a range of issues. For example, “biological carbon sequestration projects” made up 26% of the transactions in the voluntary carbon markets in 2007 (Hamilton et al., 2007). This included projects that involved storing carbon through a range of different activities:

- Afforestation/reforestation with native species: 42%
- Avoided deforestation: 28%
- Agricultural soil management: 16%
- Afforestation/reforestation in plantation monocultures: 13%
- Other biological sequestration (such as wetlands preservation): 0.1%

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For companies buying voluntary credits for the sake of public relations, the tangible nature of conserved land and general understanding of trees in the carbon cycle adds to their appeal – so-called “charismatic carbon.”

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## **OVERVIEW OF U.S. LEGISLATIVE INITIATIVES AND THE ROLE OF FOREST CARBON**

In order to illustrate the issues to be addressed as part of any U.S. national policy on forest carbon, this chapter focuses on three legislative initiatives and four sets of carbon market rules. Each is described briefly below (as of June 2009). We then dig more deeply into how each of these initiatives addresses key issues facing forest carbon offsets and public funding for forest carbon sequestration efforts.

### **Legislative initiatives covered**

Three legislative initiatives are considered: one regional and two national.

#### *Regional Greenhouse Gas Initiative (RGGI)*

RGGI is a multi-state, mandatory cap and trade program to reduce CO<sub>2</sub> emissions from electricity generation in the northeastern U.S. (RGGI, 2009). It was established in 2005 by the governors of seven states in the Northeast and Mid-Atlantic regions and has since expanded to include 10 states.

RGGI began in 2009 as the first mandatory cap and trade program for GHGs in the U.S. Its objective is to reduce CO<sub>2</sub> emissions from the regulated energy sector by 10% from 2009 to 2018. It starts by setting a regional cap to stabilize emissions from 2009 to 2014 and then reducing the cap by 2.5% each year until 2018. RGGI's first three-year compliance period started in January 2009. The program is expected to cap CO<sub>2</sub> emissions at 188 million short tons to the end of 2014. The first auction of RGGI emission allowances was held in September 2008. All the allowances were sold at a price above the auction reserve price, selling for \$3.07 per ton (RGGI, 2008b).

Offsets serve as a limited alternative compliance mechanism for regulated facilities under the RGGI program (RGGI, 2008a, § XX-10). Five types of offsets are defined in the rule as qualifying for use in the program (see discussion below) (RGGI, 2008a, § XX-10.3.a.1). While the amount of emissions that can be offset is limited, the use of offsets can be expanded if the price of emission allowances rises beyond \$7 per ton. As such, it remains to be seen what the future role and size the offset market will be under the RGGI program.

#### *Boxer-Lieberman-Warner Climate Security Act of 2008*

The Climate Security Act was first introduced before the 110th Congress in October 2007 by Senators Joe Lieberman and John Warner (S. 2191, 2007). It was later amended in May 2008 by Senator Boxer (S. 3036, 2008). The bill proposed the establishment of a market-based cap-and-trade program for GHG emissions in the United States. The cap would cover emission from U.S. electric power, transportation, manufacturing, and natural gas sources that together account for 87% of U.S. greenhouse-gas emissions. The bill also included substantial funding for agriculture and forestry programs that cut emissions, but may not qualify as offsets.



### *Waxman-Markey Clean Energy and Security Act of 2009*

The American Clean Energy and Security Act of 2009 (ACES) is a climate change and energy bill presented by Chairman Henry Waxman of the Energy and Commerce Committee and Chairman Edward J. Markey of the Energy and Environment Subcommittee. Their “discussion draft” went to the U.S. House Committee on Energy and Commerce in March 2009 and a substantially revised version was passed by the full House in June 2009. The ACES aims to “create jobs, help end our dangerous dependence on foreign oil, and combat global warming.” To meet these goals, the legislation has four titles:

- A clean energy title that promotes renewable sources of energy, carbon capture and sequestration technologies, low-carbon fuels, clean electric vehicles, the smart grid, and expanded electricity transmission;
- An energy efficiency title that aims to increase energy efficiency across all sectors of the economy, including buildings, appliances, transportation, and industry;
- A global warming title that places limits on emissions of GHGs; and
- A transitioning title aimed at protecting U.S. consumers and industry while promoting green jobs during the transition to a clean energy economy (H.R. 2454, 2009).

Table 1 proves an overview of how the three U.S. legislative initiatives address forests as part of their climate mitigation strategies:

**Table 1** Role of Forests in major U.S. legislative proposals/programs

Proposal/Program	Rules for Forest Offsets	Public Funding Programs
RGGI	Detailed requirements for afforestation offsets set forth in RGGI model rule.	None provided.
Boxer-Lieberman-Warner	Would accept forest offset projects registered in CAR, 1605b, RGGI and CCX following their respective guidelines. No further details provided.	4.25% to 4.5% of allowances issued until 2050 to be sold by the USDA to fund a rewards program for forestry and agricultural emission reduction/storage activities. Funds also provided for ecosystem adaptation to climate change.
Waxman-Markey	USDA will establish a GHG Reduction and Sequestration Advisory Committee to determine guidelines for forestry offsets. They will give “due consideration” to existing methodologies.	EPA to sell a declining percentage (from 5% to 2%) of annual allowances to generate funds for incentivizing reduced deforestation in developing countries.

Source: Data from RGGI, 2008a; U.S. Congress, 2008; U.S. Congress, 2009.

While most legislative initiatives specify the types of forest systems covered and the funding methods proposed, few (other than RGGI) offer many details on forest carbon accounting or quality assurance issues. Rather, many refer to existing efforts in the voluntary carbon markets to define rules and establish registries for the types of carbon offsets that will be accepted for trading.

### **Rules for offsets sold in the voluntary markets**

In addition to the three legislative initiatives described above, this chapter analyzes four sets of rules in the U.S. voluntary carbon markets: the Climate Action Reserve (CAR), the Voluntary Carbon Standard (VCS), the Chicago Climate Exchange (CCX), and the U.S. Department of Energy's Reporting Program 1605(b). These initiatives serve as reference points for future regulatory efforts given that they provide both applied market experiences, as well as examples of rules formulated in the complex political realities of the U.S.

The protocols for offset projects across these groups vary significantly. This is partially due to the complicated nature of accounting for forest projects to ensure that sequestration is real and based in environmental integrity. It also is indicative of the regional priorities in climate change policy. The diverse frameworks for registering forest projects attest to the complexity of designing a forest carbon accounting system.

#### *Climate Action Reserve (CAR)*

The CAR is a private non-profit organization originally formed by the State of California. It is the parent organization of the California Climate Action Registry, a body that registers and tracks voluntary greenhouse gas emission reduction projects. CAR's purpose is to establish regulatory-quality standards for the development, quantification and verification of GHG emissions reduction projects (CAR, 2009a). For projects meeting its rules, carbon offset credits known as Climate Reserve Tonnes (CRTs) are issued for the emission reductions generated. Sales and ownership of CRTs are tracked over time in a publicly accessible registry system (CAR, 2009a). The rules set by CAR are likely to have a major influence on defining the offsets that will qualify under any cap and trade program that the state of California may adopt under its climate legislation.

#### *Voluntary Carbon Standard (VCS)*

The Voluntary Carbon Standard was established by the World Economic Forum and the International Emissions Trading Association in 2005 ([www.ieta.org](http://www.ieta.org)). It is a global program working to provide a standard and a mechanism for approval of credible voluntary carbon offsets across multiple voluntary programs. It has established the voluntary carbon unit (VCU) as a means of providing tradable offset credits. The VCS focuses on a chain of ownership through its multiple registries and publically available central project database, striving to prevent voluntary offsets from being used twice. The VCS has approved the offset rules under the UNFCCC's Clean

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While most legislative initiatives specify the types of forest systems covered and the funding methods proposed, few (other than RGGI) offer many details on forest carbon accounting or quality assurance issues.

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Development Mechanism and Joint Implementation Program, as well as the Climate Action Reserve Program, as meeting its rigorous registry criteria.

### *Chicago Climate Exchange (CCX)*

The Chicago Climate Exchange was launched in 2002 as a voluntary greenhouse gas (GHG) emission cap and trade system for North America (CCX, 2009a; Kollmuss et al., 2008). Although participation in the CCX cap and trade program is voluntary, once entities elect to participate and commit to emission reduction targets, compliance is legally binding. Members can comply by cutting their emissions internally, trading emission allowances with other CCX members, or purchasing offsets generated under the CCX offset program. There are no limits on the use of offsets for compliance with parties' emission reduction commitments.

### *U.S. Department of Energy's Reporting Program [1605(b)]*

Section 1605(b) of the Energy Policy Act of 1992 established a program on the Voluntary Reporting of Greenhouse Gases (USDOE, 2007). Its purpose was to encourage corporations, government agencies, non-profit organizations, households, and other private and public entities to submit annual reports of their greenhouse gas emissions, emission reductions, and sequestration activities. Included are rules on reporting emissions and emission reductions from forest and other land-based activities (USDOE, 2007).

Taken together, these three legislative initiatives and four sets of offset rules offer a range of options for including forest carbon in future U.S. legislation – from market-based approaches involving offsets for emission reduction projects, to public funding for forestry activities. The implications of the different approaches taken are explored below.

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The diverse frameworks for registering forest projects attest to the complexity of designing a forest carbon accounting system.

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## **MARKET APPROACHES: INCLUDING OFFSETS FROM FOREST CARBON PROJECTS IN U.S. POLICY**

This section presents a guide to the treatment of forest carbon projects under the legislative initiatives and carbon market rules covered in this analysis. Each of the following aspects is considered:

- *Forest Carbon Accounting:* Forest systems; carbon pools; project sites; and baselines
- *Quality Assurance:* Additionality; monitoring and verification; leakage; and permanence

### **Forest carbon accounting**

#### *Scope of forest systems allowed*

As shown in Table 2, while most of the legislative initiatives allow carbon offset credits generated from afforestation and reforestation projects, there has been much debate over whether to include managed forests, conservation forests, harvested wood

products and other forest systems as approved carbon sinks. Part of the concern over expanding offset eligibility is the ability to track carbon storage effectively. This is especially true in the case of harvested wood products, given the uncertainties associated with their end use once they leave the forest (i.e. incorporated into a solid wood product, burned, decayed, etc.; see Chapters 12 and 13 of this publication).

**Table 2 Forest systems allowed as offsets in U.S. legislation**

Legislative Proposal/Program	Eligible Domestic Forest Offset Projects	Other Eligible Offset Projects
RGGI	Afforestation	Landfill methane capture and combustion; Sulfur hexafluoride (SF <sub>6</sub> ) capture and recycling; End-use fossil fuel (natural gas, propane, and heating oil) energy efficiency; Methane (CH <sub>4</sub> ) capture
Boxer-Warner-Lieberman	Afforestation/reforestation; Forest management.	Agricultural and rangeland sequestration (e.g. altered tillage, winter cover cropping, conversion of cropland to grassland, fertilizer reduction); manure management (waste aeration, methane capture and combustion). List subject to revision by EPA.
Waxman-Markey	Afforestation/reforestation; Conservation forestry; Improved forest management; Reduced deforestation; Urban forestry; Agroforestry; Management of peatland; Harvested wood products.	Agricultural, grassland, and rangeland sequestration (e.g. altered tillage practices, winter cover cropping, reduction of fertilizer, etc.); Manure management and disposal (e.g. waste aeration, biogas capture, substitute for commercial fertilizer).

Source: Data from RGGI, 2008a; U.S. Congress, 2008; U.S. Congress, 2009.

Under RGGI, afforestation is the only approved forestry-related offset project type (RGGI 2008a, § XX-10.3.a.1). The Boxer Amendment (S. 3036) to the Lieberman-Warner bill (S. 2191) expands on afforestation by also allowing offsets from reforestation of lands not forested as of October 18, 2007, and forest management for increased stand volume (hence increased carbon storage). However, this list would be subject to further revision by the Environmental Protection Agency.

The Waxman-Markey Bill passed by the U.S. House of Representatives in June 2009 contains the most extensive list of forestry projects as approved offset types and includes forest projects that are not commonly accepted as forest offsets (H.R. 2454, 2009, § 733). For example, urban forestry, harvested wood products, and peatland management are currently included as potential offset projects. While this list is subject to further revision, the bill requires the Secretary of Agriculture to publish within one year an official list of offset project types that will be allowed under a federal system. Acknowledging that there is still some uncertainty as to what offset projects are truly verifiable, additional, and permanent, the bill requires this list of offset practices to be revised every two years by the Secretary.

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Taken together, these three legislative initiatives and four sets of offset rules offer a range of options for including forest carbon in future U.S. legislation – from market-based approaches involving offsets for emission reduction projects, to public funding for forestry activities.

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As Table 3 highlights, all of the market rules permit afforestation. Historically, reforestation and afforestation have been the favored forest carbon project types, partly due to the ease in calculating baselines and additionality. More recently, CAR, CCX and the DOE 1605(b) have also incorporated sustainably managed forests as approved forest offset systems.

**Table 3**

Offset Rules	Scope: Forest Systems
CAR	Reforestation; Managed forests; Forest conservation. All must utilize natural forest management practices
VCS	Afforestation/reforestation (A/R); Improved forest management (IFM); Forest conservation (REDD)
CCX	Afforestation; Improved forest management; Harvested wood products; Rangeland soil carbon management
DOE 1605b	Afforestation; Reforestation; Agroforestry; Forest conservation; Sustainability managed forests; Urban forestry; Short rotation biomass; Harvested wood products

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

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While most of the legislative initiatives allow carbon offset credits generated from afforestation and reforestation projects, there has been much debate over whether to include managed forests, conservation forests, harvested wood products and other forest systems as approved carbon sinks.

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The selection of forest system types permitted in an offset regime will prove to be an important decision. The portfolio of approved forest offset projects must ensure that offset supply will be sufficiently large to assure liquidity in the market (i.e. ease of buying and selling offset credits without causing a significant movement in price). At the same time, carbon storage and uptake in these forest systems must be efficiently and accurately quantified and verifiable. This becomes increasingly complicated with forest types such as managed and conservation forests where it is difficult to accurately quantify baselines or flows of carbon storage over time (e.g. change in carbon stores post-harvest) compared to afforestation and reforestation projects (that start with little or no stored carbon).

### *Carbon pools*

Carbon pools are the parts of a forest system in which carbon is stored. These pools may include above-ground biomass, below-ground biomass, soils and wood products, among others.

To date, all offset market rules include, and most require,<sup>1</sup> that above-ground biomass be used as an approved carbon pool (Table 4). However, there is still much debate around whether to account for carbon stored in below-ground biomass, soils, and harvested wood products in an offset regime.

The ultimate decision on what pools should be included will involve a balance of costs and benefits. For example, soils are known to be a significant carbon pool. However, soil carbon is highly variable depending on site conditions and land use history. This variability increases the amount of sampling required and, therefore, the cost to accurately estimate soil carbon stocks (see Chapter 2 in this volume). If the quantification and verification of carbon pools is too expensive, it may not make

<sup>1</sup> Note: Required in all except DOE 1605(b) in which all pools are optional.

sense to include these pools. However, if a higher price is obtained in the carbon markets for more accurate measurements of carbon pools, then it will make sense for carbon developers to incur these increased costs. As such, carbon pools that can be accurately quantified may eventually be included, regardless of monitoring costs.

**Table 4**

Offset Rules	Carbon Pools
RGGI	Above-ground living tree biomass; belowground living tree biomass; soil carbon; dead biomass (unless pool is at or near zero, in which case it is optional); aboveground non-tree biomass (optional)
CAR	Above-ground living biomass; belowground living biomass; standing and lying dead biomass; litter (optional); soil (optional); wood products (optional)
VCS	Above-ground tree biomass (non-tree excluded); belowground biomass (A/R required); deadwood (IFM required); harvested wood products (IFM/REDD required)
CCX	Above-ground living trees; below-ground living biomass; soil carbon
DOE 1605b	(All optional) Above-ground living biomass; belowground living biomass; standing and down dead trees; below-ground dead trees; litter; soil carbon; harvested wood mass

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

In addition to the question of what carbon pools should be included in the rules for allowable offsets, a different question remains around what carbon pools should be measured in order to ensure the environmental integrity of forest carbon projects. As it currently stands, above-ground biomass is the only carbon pool that is approved and required in all registry systems. However, it is possible that in order to avoid deleterious ecosystem effects, other carbon pools should also be accounted for. For example, afforestation of inappropriate sites can result in an increase in above-ground carbon stores, but depletion of below-ground soil carbon for an overall net loss of carbon (Paul et al., 2002). Moreover, some afforestation or reforestation projects may result in water quantity and/or quality loss (Farley et al., 2005).

To protect against such negative ecological results, impacts on both above-ground biomass and soil carbon should be considered in afforestation projects. However, this is not currently required under any legislative or carbon market rules.

### *Spatial scale*

Another key decision in developing an offset market is determining from where offset credits can be sourced. For example, RGGI currently requires that offset credits originate from projects in one of the 10 Northeastern or Mid-Atlantic states that have signed the RGGI protocol (Table 5) (RGGI, 2008a). This rule has raised concerns over whether the RGGI offset market will be large enough to be liquid and efficient. The Massachusetts Department of Environmental Protection was sufficiently concerned about this that they decided to expand the offset project location rules to include international offset projects. They stated that insufficient offsets were available in the U.S. for facilities to achieve compliance (MADEP, 2007).

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There is still much debate around whether to account for carbon stored in below-ground biomass, soils, and harvested wood products in an offset regime.

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Table 5

Proposal/Program	Offsets from Forests in the US	Offsets from Forests outside the US
RGGI	Offsets are limited to 3.3% of a facility's emissions, but amount can be increased when allowance price exceeds \$7. Afforestation only type of forest-based offset recognized.	Not accepted.
Boxer-Lieberman-Warner	15% of emissions can be offset. Forest-based offsets allowed, including afforestation, reforestation, forest management, as well as carbon storage in agricultural and range lands.	International forest offsets are allowed, but can only be used if less than the full 15% of allowable domestic credits are used. Offsets from avoided deforestation efforts in other countries allowed.
Waxman-Markey	2 billion tons of GHG emissions per year can be offset. Half of these offsets, or 1 billion tons worth, can come from domestic agricultural or forestry projects.	50% of total offsets (1 billion tons) can come from int'l offsets. This may be extended to 1.5 billion tons of domestic offset. Market is limited.

Source: Data from RGGI, 2008a; U.S. Congress, 2008; U.S. Congress, 2009.

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To protect against such negative ecological results, impacts on both above-ground biomass and soil carbon should be considered in afforestation projects. However, this is not currently required under any legislative or carbon market rules.

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In slight contrast, the Boxer-Lieberman-Warner bill favors domestic offsets by only allowing the use of international offsets when compliance could not be met by domestic projects alone (S. 3036). The Waxman-Markey bill also favors domestic offsets by requiring five international offsets for every four tons of emissions offset, i.e. by holding 1.25 offset credits in lieu of an emission allowance (H.R. 2454, 2009, § 722). Offsets can be used by covered entities to satisfy a percentage of their compliance obligation, up to a total of approximately 2 billion tons of CO<sub>2</sub>e per year (H.R. 2454, 2009, § 722). Up to 50 percent of these offsets (or 1 billion tons per year) may come from domestic forest and agricultural offsets or from international reduced deforestation projects. If supplies of U.S. offsets prove to be limited, the Secretary of Agriculture may permit an increase in the number of international offsets to up to 1.5 billion tons, but the overall 2 billion ton limit on offsets will still hold.

### *Baselines*

Baselines are a quantitative assessment of the likely amount of carbon stored (or emissions produced) if the offset project had never taken place – such as what would have happened as part of “business-as-usual” if the offset developer had not taken steps to increase carbon sequestration (Pfaff et al., 2000). Baselines are critical measurements, as most carbon markets only grant offset credits for the extra or “additional” carbon stored by the project.

As such, the methods for establishing baselines are an important policy choice, as they dictate what forest-based activities are incentivized and qualify for carbon offsets. Baselines may be calculated by extrapolating from recent regional trends, current growth rates, existing project emissions or other quantitative measures. The most common methods used for establishing forest carbon baselines are “business as usual,” “base year” or “without-project.” Table 6 summarizes how baselines are calculated under RGGI and the carbon market rules covered in this chapter.

Table 6

Offset Rules	Baseline
RGGI	Base year approach: net increase in carbon relative to the base year (often the year prior to beginning the offset project)
CAR	Business as usual; Conservation-characterized either by a site-specific immediate threat or county conversion trends [based on Fire and Resource Assessment Program (FRAP) data]
VCS	Business-as-usual baseline. With IFM, baseline is “most likely land use in absence of project.” Three means to establishing REDD baseline depending on type of REDD activity. CAR and Clean Development Mechanism (CDM) baselines accepted.
CCX	Base year approach: net increase in carbon relative to previous year.
DOE 1605b	Base year approach: net increase in carbon relative to previous year.

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

The “business as usual” (BAU) scenario establishes a project baseline based on estimates of future emissions of a project, in the absence of carbon offset policy and without any commitments to carbon reduction (Pfaff et al., 2000). Essentially, the BAU baseline relies on projections of project-specific carbon sequestration and storage if the project proceeded untouched by carbon policy or offset credits. This approach is used by CAR and VCS for afforestation, reforestation and forest management projects (CAR 2009b § 6; VCS 2009, §3.1).

The base-year (BY) approach to establishing a baseline is similar to BAU in that the baseline is based on current project emissions and carbon storage in the absence of carbon offset policy. However, BY does not require developers to project future trends in the project’s carbon sequestration and storage. Rather, it chooses a base-year (often the year prior to beginning the offset project) to serve as the baseline and from which all “additional” carbon is measured (RGGI 2008a, § XX-10.5.c.4). So if a forest owner was planning to leave his or her plantation to grow for the next ten years without harvesting it, they would receive no offset credits for additional carbon sequestered under a BAU approach, unless they did something above and beyond normal operations. However, under the BY approach, the forest owner would be eligible for credits for all carbon sequestered by the plantation in the next ten years that is above the initial base year, regardless of the forest owner’s original intent. The BY approach is used by CCX, DOE 1605(b) and RGGI (CCX, 2009c; USDOE, 2007, § 2.3; RGGI, 2008a, § XX-10.5.c.4).

The concern with using the business-as-usual or base-year baseline is that they tend to reward project developers that have not previously adopted carbon sequestering or storage practices (Fenderson et al., 2009). For example, land managers that have been clearcutting forests would have a lower baseline than those who had historically managed their forests according to an ecologically sensitive selective harvesting regime or with longer rotations. Despite the fact that the latter’s project could sequester the same amount or possibly more carbon over the lifetime of the project, they would essentially receive fewer offsets credits than the first land manager as their baseline began at a higher value. In essence, this creates a system that penalizes good actors that have already incorporated silviculture practices that increase carbon sequestration into their forest management regime.

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The methods for establishing baselines are an important policy choice, as they dictate what forest-based activities are incentivized and qualify for carbon offsets.

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“Business as usual” and base-year approaches differ from baselines established by a “without-project case” method. The “without-project” case approach can either establish the baseline according to the carbon stored under the previous land use system (prior to the forest carbon project) or based on regional trends (from forest inventory data). Integrating a regional average data baseline such as the methodology used in the 1605(b) guidelines (USDOE, 2007, § 2.3) establishes baselines based on general land use practices in the project region. This type of baseline works well for forest offset projects that occur in regions with low forest density and a high threat of agricultural conversion or sprawl. They also reduce the costs of calculating the baseline.

### Quality assurance

In addition to the basic rules on accounting for carbon stored in forests discussed above, any future U.S. policy allowing forest carbon offset projects will need to ensure that the quality of the offsets is high enough to justify their use to meet emission reduction requirements. In doing so, the policy will need to address the quality assurance issues of additionality, monitoring and verification, leakage, and permanence.

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### *Additionality*

Offsets credits are granted only when an offset project’s activities (i.e. avoiding deforestation, lengthening rotations, reforestation previously cut sites, etc.) are considered ‘additional’ to those that would have occurred in any event (i.e. those reflected in the baseline scenario). Different approaches are used to demonstrate additionality across various rules for the carbon markets, such as direct measurement of the additional carbon sequestered, removal of barriers, performance beyond that required, and/or intent (Table 7).

**Table 7**

Offset Rules	Additionality
RGGI	Must be actions beyond those required by regulations or law. No credits for electric generation within RGGI states. No funding from any system or customer benefit fund. No credits or allowances awarded under any other mandatory or voluntary GHG program.
CAR	Any net increase in carbon stocks caused by the project activity relative to business-as-usual (BAU) baseline. Baseline estimates must reflect legal, physical and economic factors that influence changes in carbon stocks on project.
VCS	Proved through regulatory, economic or technology factors. Project must not be mandated by law and must face a barrier (technological, investment or institutional) that demonstrates that it would not occur otherwise.
CCX	All changes in carbon store after base year are considered additional.
DOE 1605b	Not specifically required. All changes in carbon store after base year are considered additional.

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

### *Monitoring and verification*

While some offset regimes require that projects be monitored on an annual basis

(CCX, 2009c; DOE, 2007, 1605(b) § 1, CAR, 2009b, § 8), others only require periodic reviews on a 2, 5 or 10-year basis. For example, RGGI requires that overall carbon stocks be assessed in afforestation projects at least every five years (Table 8) (RGGI, 2008a, § XX – 10.5.c.5).

**Table 8**

Offset Rules	Monitoring and Verification
RGGI	Validation through an accredited independent verifier.
CAR	Direct sampling of required carbon pools at the beginning (year 1) and end of 5-year certification intervals. Third party verifier recommended. Minimum confidence interval varies depending on entity. Reported data must be within 15% of certifier findings.
VCS	Project monitoring and ex post calculation of net GHG emission reduction required. Project monitoring should include monitoring of project implementation, land use change and carbon stocks. Ex ante accounting system, but when there is low precision then calculations should be revised based on ex post monitoring.
CCX	Validation through a CCX-accredited verifier. Small projects may use either direct measures or CCX-approved default tables.
DOE 1605b	Changes in carbon stocks are accounted for by periodic inventory and reporting. Default tables used for region, species, management intensity, productivity class. If negative balance (carbon stock losses), the losses are reported in ELA documents and the entity cannot register additional reductions. Monitoring over a 5 year period.

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

Likewise, certain initiatives require direct sampling of carbon stocks (CAR, 2009b, Appendix A; RGGI, 2008a, § XX-8), while the DOE's 1605(b) protocol estimates carbon stock based on tables for region, forest type and age (USDOE, 2007 § H). While the results from direct sampling are more robust than estimates, they often require third party assistance and/or verification, and thus the costs are higher for landowners.

While the carbon calculation default tables in the DOE's voluntary 1605(b) reporting are a simple and inexpensive approach (USDOE, 2007 § H), using them may raise concerns regarding accuracy and environmental integrity since the uncertainties surrounding any individual project can be high. In addition, this methodology may not be correct for calculating all forest carbon pools. For example, the DOE 1605(b) recommends something called the flow approach for estimating changes in soil carbon (USDOE, 2007 § H). It also provides a detailed format for estimating carbon captured and stored in harvested timber products. As a result of concerns regarding harvested forest product quantification methodologies, neither RGGI nor CCX has moved to offer credits for wood products.

### *Leakage*

Most forest carbon accounting regimes attempt to incorporate indirect impacts. This is to ensure that a forest sequestration project in one location does not result in increased logging and higher emissions in another region. Leakage is the unanticipated loss or gain in carbon benefits outside of the project's boundary as a

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result of the project activities. It is perhaps one of the most difficult items to measure, especially considering that it is often unintended and not under the control of the offset project developer.

Leakage can be divided into two types: activity shifting and market effects (Brown, 2009). Activity shifting is primary leakage – it occurs when the activity causing the carbon loss in the project area is displaced outside the project boundary (e.g., preventing deforestation in the project area may send the deforestation elsewhere). One difficult question to address with primary leakage is how large the “carbon shed” of the offset project should be. If the area of project influence is of manageable scale, primary leakage could potentially be addressed by establishing leakage prevention activities (e.g. alternative community development strategies) or including a buffer pool (setting aside a percent of the credits generated to cover leakage) (Hamilton et al., 2009).

Secondary leakage can occur as a result of market effects. Market effect leakage occurs when project activities change the supply and demand equilibrium. For example, if an offset project reduces the supply of wood products, it may cause an increase in forest logging in other regions to meet demand. Secondary leakage is difficult to monitor as market transactions are not always transparent. Moreover, market effects may occur at a regional, national and/or international scale.

While most guidelines for offset regimes mention the importance of addressing leakage, none of the U.S. regimes covered in this chapter include concrete measures to address this concern.

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**Table 9**

<b>Offset Rules</b>	<b>Monitoring and Verification</b>
RGGI	There are no guidelines for addressing leakage.
CAR	Activity-shifting leakage (within entity boundaries) assessment is required; Worksheet provided for estimating leakage due to change in forest management that results in a decrease in harvested wood products (thus displacing harvest to other forests). Registry pursuing approaches for quantifying leakage.
VCS	VCS provides a table of adjustments to be made to account for offsite leakage. Project development must demonstrate that there is no activity shifting or leakage within their operations – i.e. on lands outside the project, but within their management control.
CCX	Must verify that there is no internal leakage. There are no guidelines for addressing external leakage.
DOE 1605b	Small emitters must provide that reductions are not likely to cause increases elsewhere in entity (internal leakage). No requirements for external leakage.

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

### Permanence

Permanence is the main technical issue that differentiates forestry-based projects from many other emission-reducing projects (Richard et al., 2006). The concern revolves around the length of time for which carbon will remain stored in the forest and the possible loss of carbon stocks either naturally (e.g. decomposition of ephemeral tree tissues; respiration), on purpose (e.g. timber harvests) or as a result of natural disasters (Aukland and Costa, 2002). For example, while CAR considers permanence of forest projects on a 100 year basis, CCX only requires forest carbon offsets to be secured for 15 years (Table 10) (CAR, 2009b, § 7; CCX, 2009b).

**Table 10**

Offset Rules	Monitoring and Verification
RGGI	A legally binding permanent conservation easement is required.
CAR	100 year period required. Developer must insure against reversal by having buffer pool of credits and or perpetual conservation easement on the land.
VCS	An accounting method must be employed that deals with non-permanence issue from project start. The VCS approach for addressing non-permanence is to require that projects maintain adequate buffer reserves of non-tradable carbon credits to cover unforeseen losses in carbon stocks. The buffer credits from all projects are held in a single pooled VCS buffer account.
CCX	Landowners must sign contract with their aggregators attesting that the land will be maintained as forest for at least 15 years from the date of enrollment in CCX. All issuance to A/R projects shall require placement of 20% of earned Exchange Forestry Offsets in a Forest Carbon Reserve Pool.
DOE 1605b	Permanence not seen as an issue because the periodic inventory and annual reports reflect changes in net carbon flows. If the effects of natural disturbances can be separated from other causes in carbon pools, the estimated changes should not be deducted from the annual estimate for the entity.

Source: Data from RGGI, 2008a; CAR, 2009b; VCS, 2009; CCX, 2009b; and DOE, 2007.

Approaches proposed for addressing issues of permanence in forest offset projects include:

- Discounting the number of credits allowed from forest offset projects (so as to create a pool of unused credits to help cover any future increases in emissions) (VCS);
- Placement of a perpetual forest easement on the project site (RGGI, CAR); and/or
- Designing formal insurance contracts that provide buffer credits in the event of a loss (CAR, CCX, VCS).<sup>2</sup>

<sup>2</sup> In order to hedge the risk of carbon delivery failure (due to natural disaster, improper management, etc.), insurance contracts can be used to ensure there are “buffer” credits from forests that are sequestering carbon, but that are not currently accounted for in offset crediting. These buffer credits can either be from the same site as the offset credits being traded or potentially from a “buffer forest” pool.

### PUBLIC FUNDING FOR FOREST CARBON

In addition to the carbon markets, various federal programs have the potential to incentivize forestry practices that increase carbon sequestration. One approach is to

implement more carbon-sequestering forestry practices on federal lands. Another is to provide technical and financial assistance on forest management practices to private landowners. A third is to offer tax incentives to encourage carbon-sequestering forestry practices by private landowners.

In addition to these existing federal programs, the two climate bills propose to use a portion of the proceeds from auctioning emission allowances to fund a range of activities related to forest carbon (Table 11).

**Table 11**

Proposal/Program	Public Funding Programs
RGGI	None provided.
Lieberman-Warner	4.25% of total allowances issued from 2012 to 2030 and 4.5% of those from 2030 to 2050 will be sold by the USDA to fund a rewards program for forestry and agricultural emission reduction/storage activities. Funds also provided for ecosystem adaptation to climate change.
Waxman-Markey	EPA to sell a percentage of annual allowances and use funds to incentivize reduced deforestation in developing countries: <ul style="list-style-type: none"> <li>· 2012-2025 – 5% per year</li> <li>· 2026-2030 – 3% per year</li> <li>· 2031-2050 – 2% per year</li> </ul>

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For example, the Boxer-Lieberman-Warner bill allocates 4.25% of the total allowances issued from 2012-2030 and 4.5% of the allowances from 2030-2050 to the Secretary of Agriculture (S.3036, 2008, Title III and IV). Proceeds from the sale of those allowances are to be used to establish a program that rewards entities in the agricultural and forestry sectors for achieving real, verifiable, additional, permanent, and enforceable reductions in emissions or increases in sequestration, or for conducting pilot projects or other research. The bill also sets some requirements for research on agricultural and forestry GHG management.

Similarly, the Waxman-Markey bill requires the investment of a percentage of the quarterly strategic auction proceeds in programs that will further reduce the costs of climate policy, spur the development of advanced low-carbon technologies, grow the U.S. economy, and address unavoidable impacts of climate change (H.R. 2454, 2009, § 726). Included is funding for:

- Incentives to U.S. farmers and forest landowners to reduce greenhouse gas emissions and increase carbon storage in agricultural soils and forests;
- Green jobs training and assistance for workers to transition into the new jobs of a low-carbon economy;
- Reduction of deforestation and deployment of clean technologies in developing countries; and
- Programs to increase resilience to climate change impacts in the United States and in developing countries.

The bill also allows the EPA Administrator to set aside an additional percentage of annual allowances to incentivize reduced deforestation in developing countries:

- 2012-2025—5% each year
- 2026-2030—3% each year
- 2031-2050—2% each year

## MANAGEMENT AND POLICY IMPLICATIONS

In addition to the forest carbon accounting and quality assurance factors outlined above, there are a number of overarching topics that will need to be addressed as part of the forest policy discussion.

### Balancing public benefits against potential detriments from forest carbon projects

Proper management and/or conservation of forests represent an opportunity to sequester carbon dioxide and mitigate climate change. Moreover, these forest systems offer a multitude of other ecosystem services (e.g. water quality and quantity) (Graedel and van der Voet, 2009). The idea of making payments for these multiple services (in addition to carbon) may serve to make conservation financially attractive for landowners.

However, the focus should not be purely on the public benefits provided by forest offset projects – attention should also be paid to the potential for deleterious ecological impacts from carbon-focused forestry activities. For example, while afforestation may increase the carbon stored in a piece of previously unforested land, it is important to consider whether it is ecologically beneficial for the land to support trees. Afforestation or reforestation activities that require soil drainage or conversion of wetlands, as well as those that add stress to water-scarce areas, could create more public detriment than benefits.

### Accuracy versus simplicity in measurement/crediting

Accuracy in accounting for sequestered forest carbon varies according to scale: global, national, and project or site-based. The larger the area considered, the greater the uncertainties. National-level accounting is significantly more accurate than at the worldwide scale. It is believed that project-level accounting for sequestration and release of forest carbon can be achieved with 90% to 95% accuracy (Brown, 2009). These measurements are critical to calculating the carbon additionality of forest systems and awarding offset credits.

The accuracy vs. simplicity issue is also posed when considering different methodologies, such as for calculating carbon storage. While most market rules recommend direct sampling by a 3<sup>rd</sup> party verifier, DOE's 1605(b) recommends the use of look-up tables of forest conditions for a region, ownership class, forest type and productivity as a simpler and less expensive way to estimate forest carbon content (USDOE, 2007, § 1.I). While DOE notes that more elaborate models may be more accurate than look-up tables for specific activities or entities, it argues that they require more effort and significantly higher costs for not a lot of extra benefit (USDOE, 2007, § 1.I.2.6.2).

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Ultimately, tradeoffs between accuracy and cost will have to be made. One way to address these choices is to link accuracy and cost to the number of credits awarded, i.e. the more accurate your methodologies, the more credits you are issued (Brown, 2009).

### **Incentives: How to make a difference in land managers' decision making**

Other than specialist carbon developers, most land managers are not participating in the carbon markets. In part, this is because of the complexity of the various regimes, as well as the constant changes in rules and relatively low prices for land-based carbon compared to other land uses. The lack of standardized methodologies has limited the capacity of landowners to evaluate the feasibility of investments that utilize forest management as a tool to offset GHG emissions. Furthermore, the lack of publicly available, documented experience deters landowners from taking the risk of developing carbon offsets that might or might not find a market at a worthwhile carbon price.

Two of the major questions landowners should ask as the legislative debates move forward on incentives for managing forest land for carbon sequestration are the following:

- Does the legislation allow complementary funding for other environmental co-benefits, thereby increasing its attractiveness to land owners?
- How are timber management practices likely to be affected by each proposal? What are the major practical differences between them?

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## Chapter 18

# REDD Policy Options: Including Forests in an International Climate Change Agreement

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### EXECUTIVE SUMMARY

This chapter provides an overview of the role of tropical forests in the international efforts to negotiate a new global climate treaty. Under the existing treaty, the Kyoto Protocol and its “flexible mechanisms” – particularly the Clean Development Mechanism (CDM) – have succeeded in building a billion dollar market for emission reduction projects in developing countries. The role of forests and land use in those markets has been a major source of controversy, however. As a result, forests currently play an insignificant role in the markets for CDM credits – even though the greenhouse gas emissions from tropical deforestation are larger than those from the global transportation sector. Since the decision to include efforts to reduce emissions from tropical deforestation and forest degradation (REDD) in the 2007 Bali Action Plan, considerable attention has focused on designing a REDD program for inclusion in the next global climate agreement. The positions taken by different countries on REDD are driven by their circumstances – from those with large areas of standing forest to those with few remaining forests, from those facing rapid rates of deforestation to those engaged in reforestation. The overarching issues to be decided in developing the framework of a REDD mechanism include: the scope of the forestry activities to be covered; the scale of accounting for forestry activities and the baseline for measuring reference emissions levels; the type of financing to be provided for REDD activities; how to address fundamental issues of capacity and governance; and the consideration of co-benefits. There is some convergence around the scope of a REDD mechanism, the need to ultimately undertake activities at a national scale, the likelihood that financing will be both fund and market based, and

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the potential to implement REDD in phases. However, many contentious issues remain, including how to set baselines and accounting rules for REDD and how to incorporate governance concerns into a REDD agreement.

Despite the many different interests of the countries seeking to take part in a REDD mechanism and their different positions, it is possible to summarize what is known and what is not known about the key components of a REDD mechanism and where the debate stands on these issues as of the fall of 2009.

### **What we know**

- The scope of the REDD mechanism is likely to include deforestation, degradation, and “plus” activities, i.e. sustainable management of forests, conservation and enhancement of carbon stocks.
- Sub-national level accounting is likely to be allowed under the REDD mechanism as an interim measure while countries build technical capacity; however, there is consensus that a national level baseline must ultimately be reached. Therefore, the approach of scaling up from sub-national to national for countries that need time and investment to develop monitoring is likely.
- A hybrid financing system is likely to take shape that accommodates the varied interests and circumstances of states, as well as the needs of the different types of funders.
- The success of a REDD mechanism hinges on the ability of countries to address the drivers of deforestation in their countries; in many cases, addressing these issues will require significant investments not just in technical capacity, but in governance reforms and institutional capacity-building.

### **What we do not know**

- Which carbon pools to be included in a REDD mechanism have not been discussed in significant technical detail and will likely be worked out post-Copenhagen.
- While the majority of the proposals argue for the use of historic baselines, many include provisions for “national circumstances” or “development adjustment factors” that would be incorporated into the calculation of a baseline in some way, although at this stage most parties have not articulated a methodology for achieving these adjusted baselines.
- It is unclear how social safeguards or biodiversity standards might be incorporated into criteria or eligibility for REDD funding.

## **INTRODUCTION**

The problem of global climate change is of increasing concern to the scientific and political communities. Momentum is building toward a new, global agreement in

2009 for mitigating the increase of global greenhouse gas emissions and adapting to the warming that is already likely to happen based on historical emissions. While industrialized countries bear the major responsibility for these emissions, developing countries that are converting forests and other natural landscapes to uses such as agriculture and ranching are also contributing to the problem. Because emissions from land use change make up a significant proportion of global emissions, efforts are underway to develop a new strategy for bringing developing country emissions from land use change into the new climate treaty. Known by the acronym REDD, or Reducing Emissions from Deforestation and forest Degradation, it is an effort to generate resources for reducing emissions from forestland conversion in the tropics.

In this chapter, we review the history of forests in the climate negotiations, the key considerations in the negotiations among countries on the role of forests, and the major issues that will need to be worked out as part of a REDD mechanism. We close by summarizing what is known and not known about the potential framework for a REDD agreement based on the current status of the negotiations.

## THE ROLE OF FORESTS IN THE GLOBAL CLIMATE NEGOTIATIONS

### The basic structure of the global climate treaty

The United Nations Framework Convention on Climate Change (UNFCCC) was agreed in 1992 as a means for addressing a changing climate brought about by increased concentrations of carbon dioxide (CO<sub>2</sub>) and other greenhouse gases (GHG) in the atmosphere (UNFCCC, 1992). The UNFCCC established core principles of how climate change should be addressed and called for cooperation between states in information-gathering, study, and planning. Each year, the Conference of the Parties (COP) to the Convention meets to assess progress in achieving the goals of the treaty.

A Secretariat to the UNFCCC was established to provide support to the COP and the other institutions involved in addressing climate change at the international level. In addition, the UNFCCC set up two subsidiary bodies, one to provide scientific and technical advice (SBSTA) and the other to work on implementation of the treaty (SBI). SBSTA's work includes advice on technical methodologies, such as accounting for carbon in forests. The SBI's efforts include reviewing the financial assistance given by industrialized (Annex I<sup>1</sup>) countries to developing (non-Annex I) countries, as well as assessing the national emissions inventories submitted by parties. In addition, periodic reviews of the science of climate change are conducted by the Intergovernmental Panel on Climate Change (IPCC), a joint project of the UN Environment Program (UNEP) and the World Meteorological Organization (WMO).

While the UNFCCC imposed general duties on all the Parties (some more than others), specific emission reduction commitments for industrialized countries and the methods for achieving them were agreed in 1997 with the adoption of the Kyoto Protocol. The Protocol entered into force in 2005 and expires in 2012. It establishes

<sup>1</sup> Annex I Parties to the United Nations Framework Convention on Climate Change (UNFCCC) include the industrialized countries that were members of the OECD (Organisation for Economic Co-operation and Development) in 1992, plus countries with economies in transition (the EIT Parties), including the Russian Federation, the Baltic States, and several Central and Eastern European States.

both collective and individual emission reduction commitments for industrialized (Annex I) countries. Annex I nations as a whole committed to reduce their GHG emissions to 5.2% below 1990 levels by 2012 (UNFCCC, 1997). Countries' individual targets vary according to national circumstances. For example, the Protocol requires the European Union to limit emissions to 8% below 1990 levels, while Iceland and Australia were allowed to increase their emissions by a specified amount. Non-Annex I (developing) countries were not required to make binding commitments to reduce their emissions under the Protocol (UNFCCC, 1997).

The Kyoto Protocol allows Annex I countries to meet their emission reduction commitments in two general ways: (1) through domestic action; or (2) by using one of several "flexible mechanisms." Measures to reduce domestic emissions can take many forms, such as carbon cap and trade regimes, taxes, regulatory limits, incentive programs or information requirements. The flexible mechanisms allow Annex I countries to pay other countries or organizations in other countries to reduce their emissions, rather than having to reduce domestic emissions even further. One mechanism allows Annex I countries that have reduced their domestic emissions to lower than required levels to sell some of their unused national rights to emit (Assigned Amount Units or AAUs) to other Annex I countries that are having trouble meeting their targets (so-called Emissions Trading) (UNFCCC, 1997).

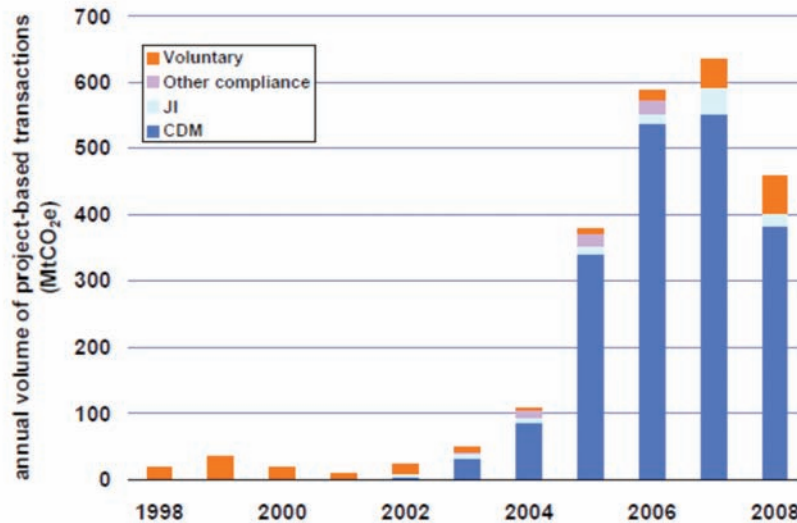
The other two flexible mechanisms under the Kyoto Protocol take place at the project level, rather than at the national level. Both allow Annex I governments or emitters, in effect, to help meet their emission reduction requirements by paying an emitter in another country to reduce its emissions instead. When the emission reduction project is located in another Annex I country, it is called a Joint Implementation (JI) project (UNFCCC, 1997). When the project is in a developing or non-Annex I country, it is done under the Clean Development Mechanism (CDM) (UNFCCC, 1997). Given that non-Annex I countries are not subject to emission reduction commitments under the Kyoto Protocol, protections were put in place to ensure that the emission reductions from CDM projects are real and deserving of credit against the commitments by Annex I countries. The CDM Executive Board (EB) was established under the UNFCCC to oversee the crediting process, from approving project methodologies to issuing tradable emission reduction credits (Certified Emission Reductions or CERs) (Paulsson, 2009). A useful source of information on the extensive rules governing the CDM program is provided in the CDM Rulebook at <http://cdmrulebook.org/>.

While the CDM has faced its share of critics (for example, see Paulsson, 2009), it has been remarkably successful in increasing the amount of private investment in emission reduction projects in developing countries. For example, since 2001 the total volume of credits under the CDM program rose from zero to a high of almost 550 MtCO<sub>2</sub>e in 2007 (Figure 1) (Capoor and Ambrosi, 2009).

In large part, this increase stems from the fact that European countries have allowed CDM credits to be recognized and traded as part of the EU's GHG Emissions Trading System.<sup>2</sup>

<sup>2</sup> For information on the EU ETS see: [http://ec.europa.eu/environment/climat/emission/implementation\\_en.htm](http://ec.europa.eu/environment/climat/emission/implementation_en.htm)

Figure 1 Annual volumes (MtCO<sub>2</sub>e) of project-based emission reductions transactions



Source: Capoor, K. and Ambrosi, P., 2009. State and Trends of the Carbon Market 2009. The World Bank, Washington, DC. Reprinted with permission.

As the international focus shifts from the implementation of the Kyoto Protocol to what will take its place when it expires in 2012, questions about the roles of carbon markets and project based credits, as well as the common, but differentiated responsibilities of industrialized and developing countries continue to pose real issues for negotiators. The debates over the role of forests in the global response to climate change reflect the difficulty of negotiating a climate agreement that seeks to balance the historical responsibility of Annex I countries with the need for all parties to undertake mitigation actions.

### Forests, the UNFCCC and the Kyoto Protocol

Land use, land use change and forestry (LULUCF) issues have traditionally played second fiddle to energy issues in the global climate discussions. This is true for a variety of reasons, including that most emissions of GHGs come from the burning of fossil fuels; forests are complex and changing systems – both storing and emitting GHGs over time; and different countries have different opportunities to include forests and other land uses as part of their response to climate change (Boyd et al., 2008).

As a result, the current discussions over options for REDD in tropical forests build on a contentious history of decisions to limit the role of LULUCF in the global climate policy instruments. Article 4 of the UNFCCC starts with commitments by all parties to: inventory the storage of GHGs in sinks (such as forests); promote processes that reduce GHG emissions in the agriculture and forestry sectors; and

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The debates over the role of forests in the global response to climate change reflect the difficulty of negotiating a climate agreement that seeks to balance the historical responsibility of Annex I countries with the need for all parties to undertake mitigation actions.

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promote the enhancement of sinks in forests and other ecosystems (UNFCCC, 1992). At the first Conference of the Parties in 1995, a pilot program of “Activities Implemented Jointly” (AIJ) projects was launched to reduce emissions, including many in the forestry sector (Boyd et al., 2008).

However, by the time negotiations over the Kyoto Protocol began in earnest, LULUCF and the role of sinks more generally had become one of the most controversial issues facing the parties (Bettelheim and D’Origny, 2002). In addition to the reasons noted above, there were concerns that allowing tradable credits from forestry projects would swamp the nascent carbon markets. The fear was that this would both delay action by industrialized countries to reduce their own emissions, as well as depress the market price for carbon credits thereby undermining the incentives for changing energy systems (Wainwright et al., 2008). Concerns were also expressed that the methodologies implemented might not be robust enough to ensure real reductions of carbon emissions and that the benefits of forest carbon projects might not accrue to the local communities living in the forests (Skutsch et al., 2007). As such, while the Kyoto Protocol allows Annex I countries to claim credit for the use of domestic sinks and expressly includes sinks in the JI Program, it is silent on the use of sinks under the CDM (UNFCCC, 1997).

The continuing debates over the roles of sinks and emissions trading contributed to the failure of the parties to reach agreement during COP-6 in The Hague in 2000 (Bettelheim and D’Origny, 2002). Soon thereafter, the Bush administration announced that it would not ratify the Kyoto Protocol. Meeting in Bonn later in 2001, the other parties agreed that while Annex I countries could use domestic sinks to help meet a portion of their emission reduction commitments, only afforestation and reforestation (A/R) projects (not avoided deforestation or forest management) would qualify for tradable credits under the CDM.

Further limits on the use of A/R projects were imposed at COP-9 in Milan in 2003. Most important was the decision to address permanence and leakage concerns by making credits from A/R projects temporary (Boyd et al., 2008). The decision to allow only temporary credits (tCERs or ICERs) from A/R projects meant that complicated, time-consuming rules had to be followed to generate a less valuable carbon commodity compared to the CERs from all other types of approved emission reduction projects.<sup>3</sup> The decision by the EU not to recognize forestry-based projects in its Emissions Trading System was another blow to the markets for A/R credits (EU Linking Directive, 2004).

The result has been that land use, land use change and forestry remain sidelined in the global carbon markets. Of the 2,148 CDM projects registered by the Executive Board as of July 30, 2009, only six were A/R projects – less than 0.3% (UNFCCC, 2009a), and agro-forestry projects made up less than 0.1% of the volume of CDM projects in 2008 (Capoor and Ambrosi, 2009).

### **REDD and the 2007 Bali Action Plan**

As recognition of the urgent need for major emission reduction has grown, forests and other land use issues are coming back into the mainstream of the global climate

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Land use, land use change and forestry remain sidelined in the global carbon markets.

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<sup>3</sup> For a detailed description of the rules governing A/R projects, see Pearson et al., 2006, *A Guidebook for the Formulation of Afforestation and Reforestation Projects under the Clean Development Mechanism*.

negotiations. Again, this is happening for a number of different reasons, including the fact that GHG emissions from deforestation are larger than those from the entire transportation sector (IPCC, 2007); that there is the opportunity to bring developing countries with large forested areas more directly into the global climate negotiations; and that the cost of reducing emissions/storing carbon in forests or grasslands is lower compared to many other mitigation options (Boyd et al., 2008).

This new round of discussions around forests started in earnest at the 2005 Conference of the Parties. Two key members of the Coalition for Rainforest Nations, Costa Rica and Papua New Guinea, introduced a proposal on reducing emissions from deforestation (RED) (Wainwright et al., 2008). Their submission suggested that a mechanism for preserving tropical forests could be financed through the carbon markets and could also provide a sustainable way forward for developing countries to mitigate their emissions from forests. Brazil proposed a quite different approach in 2006. Instead of relying on the carbon markets and private investment, Brazil's position was that reducing emissions from tropical deforestation should be paid for by a public fund (from donations by industrialized countries) that is used to create positive incentives for Non-Annex I countries to reduce their own emissions, rather than offsetting emissions from Annex I countries (Wainwright et al., 2008).

While the initial proposals put forth by Costa Rica/Papua New Guinea and Brazil focused on deforestation, countries with significant forest degradation (such as in the Congo Basin) or those with little remaining forest cover but active reforestation programs (such as India), objected to proposals focusing solely on reducing emissions from deforestation. Their position was that degradation and forest management/conservation also needed to be part of the package (REDD or REDD+) (Potvin and Bovarnick, 2008).

At the 2007 Conference of the Parties in Bali, a compromise was reached as part of the Bali Action Plan (UNFCCC, 2007). Included is a commitment to include REDD as part of the national and international mitigation actions to be undertaken under the successor agreement to the Kyoto Protocol, including provisions addressing “issues relating to reducing emissions from deforestation and forest degradation in developing countries;” as well as “the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries” (UNFCCC, 2007).

### **Forests on the road to Copenhagen 2009**

Since the Bali Agreement, the negotiations under the UNFCCC have proceeded along two tracks:

- the Ad Hoc Working Group on the Kyoto Protocol (AWG-KP).
- the Ad Hoc Working Group on Long-Term Cooperative Action (AWG-LCA).

Both working groups have held several negotiating sessions between the annual COP meetings. These meetings are an attempt to work out major conceptual issues and make progress towards developing a text for the negotiations in Copenhagen in 2009.

The bulk of the negotiations on REDD have taken place through the AWG-LCA, which is tasked with leading a “comprehensive process to enable the full, effective and sustained implementation of the Convention through long-term cooperative action” by all parties, industrialized and developing, with the goal of signing a new climate agreement at COP-15 in Copenhagen (UNFCCC, 2009b). The work of the AWG-KP, to the extent that it addresses the CDM and the role of forests in any future carbon trading regime, seems likely to raise issues similar to those in the REDD discussions – but the two groups are currently working separately.

As such, the rest of this chapter deals primarily with the party submissions that have been made to the AWG-LCA over the past two years and their implications for the more detailed structure of the REDD mechanism. However, before reviewing the details of the REDD submissions, it is important to consider the complex dynamics of international governance and its implications for expanding the role of forests in a new climate agreement.

## **INTERNATIONAL GOVERNANCE AND COUNTRY PERSPECTIVES IN THE CLIMATE NEGOTIATIONS**

### **The basic themes of international governance**

When a problem transcends an individual state's borders and affects enough people, international solutions are often developed to solve it. Some of these solutions occur between individual states in bilateral agreements or on a regional level through a variety of cooperative arrangements – such as the European Union or the North American Free Trade Agreement – or with the world community at large through international treaties such as the Convention on International Trade in Endangered Species or the Montreal Protocol on Substances that Deplete the Ozone Layer.

The system of international governance does not proceed from a single authority that possesses a mandate to govern and has the ability to enforce compliance. International law is based on voluntary agreements between states, of which treaties are the most concrete form. Obligations are articulated in these agreements, and states fulfill these obligations by implementing them within their territories or through their subjects (Brownlie, 1998).

As voluntary agreements, international treaties must be the product of discussion and consensus. It is often the case that choices have to be made between a strict system with few members, or a less robust agreement with broader participation (Speth and Haas, 2006). For instance, a state that agrees with the principles of a treaty but has few resources to enforce it effectively might be persuaded to participate if it is assured of the assistance of other states and that lapses on its part will not be punished. Moreover, as with any agreement, such a system can be affected by parties with strong interests in various outcomes, or by those that can use other resources to generate or hinder consensus. But in issues where the participation of many is as important as enforcement, a balance between the two must be struck, and the role of interest and power must be factored in the process of negotiation.

The need for balancing the different interests of many parties presents a major challenge for negotiating a climate change treaty. Because the scale of the problem of climate change is global, it is necessary to bring all states into a regulatory framework to address climate change. Operationalizing that framework, however, can be problematic. The UNFCCC itself has near-universal membership, reflecting broad agreement on its principles. But in working out specific obligations for different members under Kyoto, the interests of key emitting states were insufficiently addressed, reducing the overall effectiveness of the agreement. Some developed states felt that they were taking on the full burden of addressing climate change, while rapidly developing states were not bound to reduce their own emissions. On the other hand, many developing states emphasized their own need for economic development and that adopting emission restrictions would hamper their ability to secure the material well-being of their citizens (Hunter et al., 2007).

The UNFCCC is premised on the “common but differentiated responsibilities and respective capabilities” of states that are parties to the Convention, but it stresses that developed countries should be leaders in mitigating climate change and recognizes the vulnerability of many developing countries, particularly small island states and least developed countries (Stone, 2004). While all parties should take “precautionary measures to anticipate, prevent or minimize the causes of climate change and mitigate its adverse effects” such measures should “be appropriate for the specific conditions of each party, and be integrated with national development programs, taking into account that economic development is essential for adopting measures to address climate change” (UNFCCC, 1992) .

### **REDD as a mitigation strategy**

As the Ad-Hoc Working Group on Long-Term Cooperative Action (AWG-LCA) stated at its 5<sup>th</sup> session in Bonn in April 2009, “a REDD mechanism should be designed to accommodate differing national circumstances and respective capabilities within and between developing countries on issues relating to reducing emissions from deforestation and forest degradation, and the role of conservation, sustainable management of forests, and enhancement of forest carbon stocks” (AWG-LCA, 2009). Reducing emissions from deforestation by safeguarding the world's remaining forests can be a critical step on that path.

REDD has thus received increased attention for its potential to address the concerns of both developed and developing countries. Developed countries that bear most of the responsibility for the current global emissions see REDD as a cost effective mitigation tool that will enable them to help meet their own emission reduction targets through the sale of carbon offsets or credits. On the other hand, many developing countries see the role of REDD as a way of meeting their own mitigation goals. While the Kyoto Protocol did not require non-Annex I countries to make commitments, as a part of the Bali Action Plan, developing countries are tasked with developing “nationally appropriate mitigation actions” (NAMAs) that allow them to contribute to climate change mitigation. Many developing countries see REDD as a NAMA that can be used to meet this goal of developing national

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The need for balancing the different interests of many parties presents a major challenge for negotiating a climate change treaty.

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mitigation strategies. Whether REDD is an offset mechanism for Annex I countries or contributes towards non-Annex I mitigation goals will be a contentious issue in the Copenhagen negotiations. For example, both Brazil and Panama have emphasized that the REDD mechanism should not be a means for developed countries to meet their emission reduction commitments under Kyoto (Wainwright et al., 2008).

### Country perspectives

In order to understand the different country perspectives on REDD, it is essential to consider the differences in national circumstances with regards to forest cover and historical rates of forest loss in developing countries with tropical forests. Da Fonseca et al. (2007) have categorized countries based on their (1) remaining forest cover and (2) deforestation rate as a way of highlighting the fact that the state of the forests in a country plays a crucial role in that country's ability to benefit from – and therefore its views on – a REDD mechanism (Figure 2). Specifically, these differential circumstances underpin debates such as the type of forestry activities to be included in the mechanism, the establishment of reference emissions levels for generating carbon credits, or the scale at which activities should be undertaken.

The primary focus of REDD in the negotiations since Bali has been on providing incentives for countries to reduce deforestation rates. The countries with the highest amount of emissions from forests are Brazil and Indonesia, and both countries have moderate to high rates of deforestation (The Nature Conservancy, 2009). Therefore, the greatest potential activity for mitigating emissions from land use conversion is to reduce and eventually halt deforestation rates in these countries. In countries with significant deforestation, reducing the rate of that deforestation compared to an established baseline is the key way to generate credits or offsets for emissions reductions. However, countries with historically low deforestation rates, such as Guyana and Suriname, argue that countries that have left more of their forests standing should not be penalized for having lower rates of deforestation by having fewer REDD credits available to them. Explicitly emphasizing that focusing on reversing deforestation will leave out countries that have the highest percentage of remaining forest cover and lowest rates of deforestation, Suriname proposes that a REDD program must in fact focus on these countries and provide support for their economic development that doesn't involve cutting down their forests. Considering the situation of these countries would reduce the likelihood of leakage, and the importance as well of providing ex ante funding to avoid development pressures (SBSTA, 2008).

There is another group of countries, such as India and many of the West African nations, that have little forest cover remaining but are eager to participate in REDD through afforestation, reforestation and sustainable forest management activities. The REDD mechanism negotiated at Copenhagen must therefore balance the different national circumstances with regard to the state of forests, but also seek to develop a REDD architecture that ensures environmental integrity and ultimately reduces emissions from forests by stopping deforestation.

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Whether REDD is an offset mechanism for Annex I countries or contributes towards non-Annex I mitigation goals will be a contentious issue in the Copenhagen negotiations.

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Figure 2 Developing country circumstances classified by forest cover and deforestation rates

	Low Forest Cover (<50%)	High Forest Cover (>50%)
<b>High Deforestation Rate (&gt;0.22%/yr)</b>	<p><b>Quadrant I</b></p> <p>e.g., Guatemala, Thailand, Madagascar</p> <p>High potential for RED credits</p> <p>High potential for reforestation payments under CDM</p> <p>Number of countries:44 Forest area: 28% Forest carbon (total): 22% Deforestation carbon (annual): 48%</p>	<p><b>Quadrant III</b></p> <p>e.g., Papua New Guinea, Brazil, Dem. Rep. of Congo</p> <p>High potential for RED credits</p> <p>Low potential for reforestation payments under CDM</p> <p>Number of countries:10 Forest area: 39% Forest carbon (total): 48% Deforestation carbon (annual): 47%</p>
<b>Low Deforestation Rate (&lt;0.22%/yr)</b>	<p><b>Quadrant II</b></p> <p>e.g., Dominican Republic, Angola, Vietnam</p> <p>Low potential for RED credits</p> <p>High potential for reforestation payments under CDM</p> <p>Number of countries:15 Forest area: 20% Forest carbon (total): 12% Deforestation carbon (annual): 1%</p>	<p><b>Quadrant IV - HFLD Countries</b></p> <p>e.g., Suriname, Gabon, Belize</p> <p><b>Low potential for RED credits</b></p> <p><b>Low potential for reforestation payments under CDM</b></p> <p><b>Number of countries:11</b> <b>Forest area: 13%</b> <b>Forest carbon (total): 18%</b> <b>Deforestation carbon (annual): 3%</b></p>

Source: da Fonseca, G.A.B., Rodriguez, C.M., Midgley, G., Busch, J., Hannah, L., Mittermeier, R.A., 2007. No forest left behind. Plos Biology 5, 1645-1646. Reprinted with permission.

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The REDD mechanism negotiated at Copenhagen must therefore balance the different national circumstances with regard to the state of forests, but also seek to develop a REDD architecture that ensures environmental integrity and ultimately reduces emissions from forests by stopping deforestation.

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### KEY CONSIDERATIONS FOR A REDD PROGRAM AND HOW THEY ARE ADDRESSED IN THE MAJOR PROPOSALS

A strategy to reduce global emissions from deforestation and forest degradation must consider the diversity of interests of the states involved, and will therefore necessarily delve into many complex issues. The overarching questions to be decided in developing the framework of a REDD mechanism can be divided into several topics that have been a source of debate among both developed and developing country parties at the intersessional negotiations. The major questions can be categorized as follows:

- (1) The scope of forestry activities to be included in the REDD mechanism;
- (2) The scale of accounting for forestry activities and the baseline for measuring reference emissions levels;

- (3) The type of financing to be provided for REDD activities;
- (4) Capacity building and REDD “readiness”; and
- (5) The role of co-benefits.

### **Scope of a REDD program**

#### *Activity scope*

Over the course of the debate, there have been three ideas of what activities should be included in a program to reduce emissions from forests:

- 1) RED proposals limit included activities to only those which result in reduction in deforestation rates;
- 2) REDD includes reducing emissions from deforestation and forest degradation; and
- 3) REDD+ further expands the scope of REDD by including activities such as sustainable forest management, conservation, and enhancement of carbon stocks of forests and plantations. (Parker et al., 2008).

Some countries, such as Australia and the United States, have articulated in their submissions to the AWG-LCA that the long term goal should be for countries to do full land-based accounting, or that developing a REDD strategy should ultimately form part of a sustainable land use management plan as part of a low carbon development strategy. Recent findings from climate models suggest that if policy is focused just on energy use, the land use implications – in terms of increased deforestation for biofuels – will be huge and negative (Wise et al., 2009).

Resolving the debate about the scope of a potential REDD mechanism was the key to adopting the decision to move forward on REDD as part of the Bali Agreement at COP13 (Potvin and Bovarnick, 2008). The root of this debate is the difference between carbon sinks and sources across tropical countries. Deforestation and degradation in forests are a source of emissions, while the intact forest itself is a sink for carbon dioxide. Brazil wanted a RED mechanism that only focused on deforestation because of the fear that the inclusion of the carbon stocks of intact forests for conservation would dilute the funding sources needed for combating deforestation (Potvin and Bovarnick, 2008). As a country that has little forest cover remaining and a relatively low rate of deforestation, India proposes a common methodology that assesses: i) changes in carbon stocks and GHG emissions due to conservation and sustainable management of forests, and ii) reductions in emissions from deforestation and degradation (SBSTA, 2008). While some countries cite concerns over methodologies for going beyond monitoring deforestation, Canada asserts in its submission to COP 13 that, while methodological difficulties do exist in assessing and quantifying forest degradation, countries should not be excluded from incentives to reduce emissions from deforestation (SBSTA, 2008).

The REDD architecture is still in its developing stages as the negotiations progress, but in reviewing the most recent Party submissions on REDD to the AWG-LCA, there

is convergence around the inclusion of the “plus” activities in a REDD mechanism. However, there are still important technical and financing issues to consider within the scope of the discussion. For example, the feasibility of developing countries performing carbon accounting for activities beyond degradation may ultimately limit the initial scope of REDD in many developing countries. Additionally, the distribution and types of funding available could be contingent on the scope of the activity being implemented. Both India and Mexico have proposed financing mechanisms that are linked to the type of forestry activity being performed. Their proposals relate back to the discussion of the forest as source and sink: India proposes that market mechanisms may be suitable for activities such as deforestation (which is comparatively easier to measure), while fund-based financing may be necessary for activities that enhance carbon stocks (AWG-LCA, 2009). It will also be important, at some point, to bring together the ideas on forests from the AWG-LCA and the AWG-KP. The lessons learned from the efforts to include A/R projects (a part of many REDD+ formulations) in the CDM should be useful for discussions of REDD+. In addition, protections against double-counting projects will have to be included to ensure that any particular project is not counted by both a funding country and a host country against each of their commitments.

#### *Carbon pools/ecosystem types covered*

Because articulating a framework for REDD is still in early stages, most submissions to the AWG-LCA do not contain a great level of detail on issues such as the carbon pools that will be eligible for generating emissions reductions, i.e. above or below ground biomass, soils, or wood products (Parker et al., 2008). This is due in part to the difficulties associated with measuring carbon pools such as soil and below ground biomass (see Chapter 2, this volume). However, a few countries such as Australia and the United States propose the eventual inclusion of all forms of terrestrial carbon (grasslands, woodlands, peatlands, etc), not just forests. They advocate that these other ecosystems be phased in as science develops methods to quantify their respective carbon benefits (Ashton, 2008). While this perspective ties in with the goal of sustainable land use management, it is unclear whether or not any REDD agreement reached in Copenhagen will deal explicitly with the different carbon pools in forests. While carbon stored in above-ground biomass will certainly be included in any future REDD system, the inclusion of other carbon pools is much less certain.

#### **Baselines and accounting**

The REDD mechanism also has a myriad of options for how emissions reductions can be measured. The reference level has to define the way in which emissions reductions or carbon stock enhancements will be compared to a chosen baseline, and the scale at which carbon accounting is done will impact whether REDD is implemented at a project/sub-national or a national scale.

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### Reference emission levels

The establishment of a baseline scenario of deforestation/degradation over a defined scale in a business-as-usual scenario is the first step in accounting for REDD. This reference level greatly affects which countries will be able to generate emissions reductions and the amount of credits that will be available due to varying levels of deforestation. Because of the differences in national circumstances enumerated above, most party submissions state the need for flexibility in reference levels that allow countries with low rates of deforestation to participate in the generation of emission reductions. This flexibility is important because looking only at current deforestation rates across countries does not take into account either the historical or future drivers of deforestation that must be addressed in order to reduce emissions.

### *Historical baselines*

One potential reference scenario is measuring emissions reductions against a historical baseline. This method could be set by choosing a baseline year and comparing rates of deforestation. Many countries are proposing baseline years for their own national emissions reductions strategies, and baselines vary from 1990, 2000, or 2005 levels depending on the proposal (Olander et al., 2007). Alternately, historical reference scenarios could be developed by taking the average rates of deforestation over a defined period of time. For example, Brazil has already set its own goal of reducing its emissions against a baseline taken from its average area of deforestation from 1996 to 2005 (Carbon Positive, 2009).

While historical baselines can be relatively simple to calculate, those countries with high, current deforestation rates will gain much more from emissions reductions under a REDD program. Malaysia's proposal voices concerns that using a historic baseline will create a perverse incentive to increase timber harvests in the years before the first commitment period (Parker et al., 2008). While other countries seek to avoid this problem by not taking into account the most recent time period in developing historical baselines, the majority of proposals for REDD are not focusing on historical baselines for reasons of both equity across countries and because many developing countries lack consistent historical data on deforestation rates.

### *Projected baselines*

The Centre for International Sustainable Development Law (CISDL) advocates the use of projected baselines outlined in their "Carbon Stock Approach" (Climate Focus, 2007). They see the lack of ex ante funding (payments to countries up front for capacity building and strategy development), as a major roadblock in implementing REDD activities in developing countries that have very little capital to invest in such projects. In this scenario, a projected baseline of emissions from deforestation would be created in order to set aside a certain stock of forest carbon that is expected in the future. Countries might achieve this through using current deforestation data, information about the country's development pathway, population growth, or other data on drivers of deforestation to create predictions or use econometric models to develop future emissions scenarios.

The major criticism of the projected baseline approach is the difficulty in accurately predicting future forest trends. Many critics see room for distortion and corruption. Colombia proposes that projected baselines could be based on either an extrapolation of past trends into the future, prevailing technology or practice, or logical arguments made by activity participants based on observed trends (Parker et al., 2008).

### *Historical adjusted baseline*

At this point in the negotiations, many countries are suggesting an approach that takes into account both historical trends and national circumstances such as drivers of deforestation or development pathways that can have significant impacts on forest cover. While party submissions are vaguely worded, proposals from the United States, India, Papua New Guinea, Australia, and Norway all reference the need to develop baselines that take into account national circumstances and capacities (AWG-LCA, 2009). Norway's proposal is the most detailed. It recommends establishing reference emission levels using a formula based on inputs such as historical emissions, forest cover, and measures of per capita GNP to factor into an adjusted baseline (AWG-LCA, 2009). Setting these reference levels could be done by an expert body or technical panel that is in charge of overseeing a standardized process for setting baselines. While this type of proposal might be successful in taking into account national circumstances beyond simple measures of forest cover or deforestation rate, a methodology would still need to be developed and agreed upon by the interested parties under the Convention.

The discussion of reference levels underscores the fact that countries are likely to favor different methods of setting reference scenarios depending on their national circumstances with regards to forest area. It is unlikely that there will be a single solution for developing reference levels; developing national reference levels will likely fall to individual countries as they develop their REDD strategies, and a technical or scientific body of experts under the COP could be responsible for review and approval of country proposals.

### **Scale**

The scale at which REDD activities are implemented – project or national – is another key issue to be decided as part of the UNFCCC negotiations in Denmark. The scale of REDD poses risks to the environmental integrity of the mechanism because being able to implement REDD at a national scale requires measurement and monitoring capacity that many developing countries may lack.

### *National level*

From the perspective of ensuring the environmental integrity of REDD projects, establishing a national level baseline is a key strategy that is acknowledged by most parties as the most effective way to prevent displacement of emissions within a country from the site or from a REDD project to another area (frequently called

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leakage). National-level baselines also empower host countries to pursue a broader set of policy tools and take ownership of their projects (Angelsen et al., 2008b). India proposes a national baseline to prevent double counting and national-level leakage (SBSTA, 2008). They also propose that CDM A/R activities be debited from a national inventory for REDD accounting in order to address additionality concerns that forestry activity implemented under the CDM would be counted twice because of overlap between REDD and A/R mechanisms.

### *Sub-national/project level*

The argument for a sub-national approach presented by the Latin American coalition and Malaysia is that it 1) is easier to monitor and verify, 2) encourages investment from the private sector, and 3) could provide more direct benefits to forest-dwelling people (Potvin and Bovarnick, 2008). They argue that relying only on national level baselines is problematic as many countries lack the capacity, governance, and control of territory to effectively implement a national baseline. Use of project level baselines means that many developing countries that do not have resources to create a national carbon accounting mechanism will nevertheless have flexibility to engage in REDD activities (Parker et al., 2008). Most countries supporting a sub-national approach see it as an interim measure, acknowledging the need to eventually work towards national scale accounting.

### *Global*

The Centre for Social and Economic Research on the Global Environment (CSERGE), a research centre based out of the University of East Anglia, UK, proposes that credits should be generated relative to a global baseline as a way of eliminating international leakage (Strassburg et al., 2008). The Joint Research Centre, the European Commission's research organization, proposes an "Incentive Accounting" program where countries with emissions less than half of a global average baseline be rewarded for maintaining carbon stocks, whereas those with higher than the global average are rewarded for reducing emissions from forest conversion (Parker et al., 2008).

With regards to the scale of REDD activities, there is convergence around the idea that national baselines are the ultimate goal, but that sub-national projects should be allowed as part of a readiness or scaling up phase in order to allow countries that will not be immediately ready to do national level monitoring time to improve technical and institutional capacity.

### **Financing**

Generating financing for REDD activities at an adequate and sustainable scale is crucial to its success. This is particularly true in order to create incentives and payment systems for government actions and specific projects to reduce emissions that overcome the drivers of deforestation and forest degradation. Recognizing the varied interests and institutional capacities of states, various proposals are being discussed, ranging from creating a market for the trading of forest carbon emission

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reduction credits, establishing a public fund from contributions by states and financial institutions to pay directly for such reductions, or combinations of the two approaches.

### *Market system*

A market system to finance REDD presupposes the existence of a working cap-and-trade market for carbon credits (such as the EU's Emissions Trading System). The system will need to cover large portions of the industrial sectors of developed states (and perhaps increasing numbers of developing states with the passage of time), for which the total amount of GHG emissions will be set (the "cap") and permits for those emissions will be sold and traded between emitters (the "trade"). Under the cap-and-trade system, REDD projects could be issued offset credits once they have achieved emission reductions by protecting forests. These credits in turn could also be traded on the cap-and-trade market, and the proceeds from the sale of REDD credits would ensure continuing REDD activities.

As discussed above, many questions have been raised about the wisdom of including forest-based credits in the carbon markets. One major concern in the REDD discussions is that the price of REDD credits would be much cheaper than the cost of reducing the same amount of emissions from the energy sector, such that the economic incentives to change our use of fossil fuels would be undermined. The fear is that REDD credit prices will be so much lower than the cost of a clean energy project, for example, that it will bring large volumes of new credits into the market, thus easily meeting the demand for credits and driving the overall market price for credits lower.

As such, some proposals suggest that any tradable credits from REDD projects should not be traded at face value on any cap-and-trade market (Dutschke, 2008). Greenpeace recommends that REDD credits should be sold with a surcharge to make them more expensive and that the proceeds from the surcharge could be used to fund other institutional or capacity-building activities, over and above specific measures against deforestation and degradation (Thies and Czebiniak, 2008). Of course, proponents of market approaches argue that imposing any such additional costs on REDD projects will only limit their use, and hence, their effectiveness in mitigating or storing emissions.

While the price of REDD credits will directly benefit the entity operating the REDD project, many people believe that separate funding will also be necessary to address the broader social and political factors that contribute to deforestation, such as insecure land tenure and indigenous peoples' rights, enforcement and monitoring capabilities, economic and agricultural policy coherence, among others (Thies and Czebiniak, 2008).

### *Fund-based mechanism*

Another option for providing REDD financing is for governments, financial institutions or private entities<sup>4</sup> to contribute to a fund, which can then be disbursed to support REDD projects. How those contributions are generated can take a variety of forms. There are proposals, for instance, that the Assigned Amount Units (AAUs)

<sup>4</sup> Some carbon funds currently administered by the World Bank have voluntary private sector contributors, and there appears to be no hindrance to allowing similar arrangements for REDD funding. See its BioCarbon Fund, among others (World Bank, 2009). The amount of this "charitable" or "learning" capital, however, is relatively limited. More extensive incentives for private investors to put money into REDD projects will have to be included if substantial amounts of private investment are to be expected.

for allowable emissions from Annex I countries be auctioned and a part of the auction proceeds be used for REDD projects (AWG-LCA, 2009). In other proposals, participating governments can impose taxes within their own states or make annual appropriations and remit these to a central REDD fund. In any case, parallel to such a central fund, it would still be possible that REDD activities in one country can be directly financed by foreign governments, corporations, or financial institutions to comply with their own emissions targets through bilateral agreements such as the Amazon Fund, a fund run through the Brazilian Development Bank that is currently funded by the government of Norway (BNDES, 2009).

While most parties agree that some degree of public funding should be involved in a REDD mechanism, the major arguments against solely a fund-based mechanism is that it is unlikely to be able to generate funding at the required scale to effectively provide support for emissions reductions activities and build the capacity to monitor those activities, and that funding will not be continued long-term.

At present, some non-market funds are in place to help countries prepare for what is being termed REDD “readiness.” The Forest Carbon Partnership Facility of the World Bank and the UN-REDD Programme – run as a collaboration between the Food and Agriculture Organization (FAO), United Nations Development Programme (UNDP) and United Nations Environment Programme (UNEP) – are working with developing countries to provide support for the development of REDD strategies. The donors for these programs thus far are predominantly national governments (for example the UN-REDD Programme is funded by a \$52 million grant from Norway) (UNDP, 2008).

#### *Hybrid funding mechanism*

Noting that both market and non-market systems have their limitations, there are various combinations being explored that make the most of the strengths of each system. Some proposals suggest that the type of financing should depend on the type of action being undertaken. For example, efforts to build capacity and improve forest governance could be separated from activities that directly result in emission reductions. Contributions from funds could be used to finance the governance activities, while market-linked or direct market financing could be used to finance the actual emissions reductions. The International Institute for Environment and Development (IIED) proposes a system in which governmental transfers would focus on improving institutions and governance – improving monitoring and law enforcement, land tenure reform and indigenous rights, agricultural and economic policies, among others – while carbon markets would direct resources to people and communities to provide the incentive and support to manage forests at the ground level (Viana, 2009).

Another proposal that has gained significant support through the negotiation process leading up to Copenhagen is the phased approach enumerated by the Norwegian government. Recognizing that different countries are at different levels of institutional capacity to effectively utilize market-based financing, Norway proposes three phases of REDD that consist of: 1) a capacity building phase; 2) a scaling up

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Noting that both market and non-market systems have their limitations, there are various combinations being explored that make the most of the strengths of each system.

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phase to include government policies and measures addressing drivers of deforestation as well as demonstration activities for emissions reductions; and 3) full implementation (AWG-LCA, 2009). In this scenario, the type of funding available would depend on the phase of REDD, with initial phases being supported by non-market funds for planning, and institution- and capacity-building activities at the national level. When institutions develop sufficient capacity for monitoring and demonstrating emissions reductions, countries could proceed to a full implementation phase in which they would access the carbon market (Angelsen et al., 2008a).

While hybrid systems attempt to address the deficiencies of both market and non-market financing schemes, they also create a complicated system that will require its own bureaucracy. Transaction costs will thus increase, and target communities and programs may actually receive fewer funds. Figure 3 is a chart from IIED (Viana, 2009) that explores further the strengths and weaknesses of market and non-market (government) strategies:

**Figure 3 Strengths and weaknesses of government and market finance for REDD**

Effectiveness	Efficiency	Equity	Urgency
<b>Government</b>			
+ strong support of rainforest governments encourages sound policies	+ lower international transaction costs	+ facilitates international transfers between rich and poor countries	- slow implementation of intergovernmental funding
- limited effectiveness of government-based policies	- higher domestic costs	- favours middle-income countries	- slow implementation of government programmes
+ captures domestic leakage	+ greater incentives for governmental policies	- risk of domestic distribution inequities	
- does not capture international leakage	- greater risk of policy and governance failure		
- limited attractiveness to private funders	+ lower monitoring costs		
<b>Market-based</b>			
- weak support to encourage sound policies by rainforest governments	- higher international transaction costs for small projects	+ increases funding from market to forest communities in poor countries	+ quicker implementation of project-based activities
+ greater effectiveness of field project-based activities	+ lower bureaucracy and administrative costs	+ does not favour middle-income countries	+ quicker impacts on reduction of deforestation and degradation
- does not capture domestic leakage	- smaller incentives for governmental policies	+ smaller risk of inequitable distribution of benefits to local communities	
+ increases area of forests under protection with positive impacts on international forest leakage	+ smaller risk of policy and governance failure	- potential risk of inequitable distribution of benefits to local communities if project certification schemes are ineffective	
+ greater attractiveness to private funders	- greater monitoring costs		

Source: Viana, V., 2009. Financing REDD: Meshing markets with government funds. International Institute for Environment and Development, London. Reprinted with permission.

Market and non-market sources of financing have their strengths and weaknesses. An attempt to have the best of both will likely result in a hybrid system, and while such a system may be considerably more complex and more costly to operate, it may also be able to be more flexible in addressing the different needs of different states.

### **Capacity building and readiness**

While scope, scale of activity, and sources of financing are all critical topics to be discussed when developing a REDD mechanism, it is also essential to consider the obstacles that the limited capacity for forest and revenue management in many developing countries may present to the successful implementation of REDD. As illustrated in Chapters 14 and 15 of this volume, the drivers of deforestation in many developing countries are complex, operate across multiple spatial scales, and are not confined to the activities that directly impact forests such as logging or agricultural conversion. Additionally, forest governance, institutional capacity, and technical expertise must all be improved in order to achieve the long term goals of REDD.

There is increasing recognition that these issues of institutional capacity and national circumstance must be addressed in order for countries to more effectively engage the global REDD system. Thus, it is likely that any agreement on REDD will involve a complex system of distributing REDD benefits in order to address the differences among states in the hope of actually achieving reductions in emissions from deforestation and forest degradation. One approach with the potential to address these governance and capacity issues directly is the phased approach to REDD.

#### *Phased approach to REDD*

Considering the need to address institutional obstacles alongside activities directly linked to emission reductions, Norway proposes a phased approach to REDD proceeding first from readiness planning, then to strategy implementation (where incentives are given based on proxies for emission reductions), and ultimately to a phase where incentives are given for actual emission reductions (AWG-LCA, 2009). In each phase, obligations become more rigorous and defined.

In the first phase, funding is voluntary or bilateral in nature, depending on the country's commitment to REDD. Thus far the FCPF and UN-REDD Programme have operated in such a way that the work being done by developing countries to create REDD strategies could inform this initial phase of REDD through lessons learned and best practices in developing stakeholder buy-in and creating an implementation strategy for REDD. In the second phase, funding will be directed towards capacity building and institutional reform, financed through the auction of AAUs. Phase three will be financed through a market, with sales of forest-based credits from reduced emissions relative to an agreed baseline, dependent on an operational national GHG forest inventory. Norway's proposal also puts considerable emphasis on the role of forest stakeholders, the need to improve forest governance, respect for rights of indigenous peoples, and biodiversity conservation in the implementation of REDD (AWG-LCA, 2009).

## Co-Benefits

Many policymakers see a REDD mechanism as a method of incentivizing countries to reduce emissions while simultaneously preserving “co-benefits,” such as protecting biodiversity in forests, preserving ecosystem services, and helping to improve the livelihoods of indigenous peoples and other communities that reside in forests (Angelsen et al., 2008a).

While many of the submissions to the AWG-LCA contain language about respecting the property rights of indigenous peoples and local communities, as well as biodiversity, there have been few attempts to flesh out what these social and environmental safeguards might look like. The idea of social safeguards is prevalent in the development community and it is possible that one idea for mainstreaming biodiversity concerns into REDD is to make it part of countries’ eligibility for funding. Another issue concerning biodiversity is that of plantation forests. While most countries do not address this issue specifically, Bolivia’s proposal seeks to ensure that REDD activities do not result in the clearing of natural forests to be replaced by plantations, which might generate carbon credits as a REDD+ project but would have negative impacts on forest biodiversity (AWG-LCA, 2009).

Another serious concern in the NGO community and civil society in potential REDD countries is the impact that REDD could have on local communities and indigenous peoples who dwell in forests. One key way in which this concern can be addressed is by meaningful inclusion of all stakeholders, including local communities and indigenous peoples, in the design, development and implementation of national REDD strategies. The FCPF and UN-REDD processes thus far have required consultation and stakeholder plans to be submitted along with country proposals or strategies for creating a REDD plan. However, thus far these consultations have tended more towards education and awareness raising rather than engaging stakeholders in inclusive consultations that incorporate local perspectives into national REDD plans (Daviet et al., 2009).

Overall, the approach to considering the rights of indigenous peoples and local communities, as well as language for protection of biodiversity and ecosystem services, will need to be elaborated in order for REDD to proceed with adequate safeguards in place. The UNFCCC process may be able to address these issues by building off of the readiness processes that are currently in place. Full inclusion of all relevant stakeholders and recognition of indigenous rights, and safeguards to ensure that biodiversity is not adversely affected by REDD projects, are both essential to the integrity of the REDD mechanism.

## CONCLUSION

It is clear that the process of reaching an agreement on REDD through the UNFCCC negotiations will be complex and will require many compromises. This review of the major substantive issues and the proposals to the AWG-LCA has identified some areas in which there is convergence on the goals and objectives of the REDD mechanism,

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Overall, the approach to considering the rights of indigenous peoples and local communities, as well as language for protection of biodiversity and ecosystem services, will need to be elaborated in order for REDD to proceed with adequate safeguards in place.

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but also finds that many contentious issues remain that may require significant effort to resolve. It is important to keep in mind that the Copenhagen meeting in December 2009 is likely to put in place a general framework for REDD that will be the subject of further negotiations in the future – in the same way that the accounting rules for Annex I countries were not settled in Kyoto in 1997, but took several years to finalize (Schlamadinger et al., 2007).

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## Chapter 19

# Synthesis and Conclusions

*Mary L. Tyrrell, Mark S. Ashton, Deborah Spalding, and Bradford Gentry*

If the world wants to meet its climate mitigation goals, forests – as both a sink and source – must be included. According to the 2007 IPCC report, deforestation and land use change currently account for a third of total anthropogenic global greenhouse gas emissions. Any comprehensive climate change policy must address this issue. At the same time, forests have a significant potential to sequester carbon. Their inclusion in a climate regime could have an immediate impact.

About half of terrestrial carbon is stored in forests, which can act as a sink or a source of carbon under different conditions and across temporal and spatial gradients. Best current estimates are that the terrestrial biome is acting as a small carbon sink, most likely occurring in forested ecosystems. Boreal and temperate forests are sequestering carbon (net sinks), while tropical forests are a net source of CO<sub>2</sub> emissions due to deforestation (land use change).

Understanding the role of forests in global carbon budgets requires quantifying several components of the carbon cycle, including how much carbon is stored in the world's forests (carbon pools), gains and losses of carbon in forests due to natural and anthropogenic processes (carbon fluxes), exchanges between the terrestrial carbon and other sinks and sources, and the ways in which such processes may be altered by climate change.

This extensive review of the literature on forest carbon science, management, and policy has produced several important conclusions, and elucidated what we currently do and do not know about forests, carbon, and climate change. They are summarized here as a contribution to the current knowledge base of how to preserve the carbon stock in the world's forests and potentially maintain forests as CO<sub>2</sub> sinks into the future.

### **THE SCIENCE OF CARBON UPTAKE AND CYCLING IN FORESTS**

In order to better understand the ways in which future forests will change and be changed by shifting climates, it is necessary to understand the underlying **drivers of**

**forest development** and the ways these drivers are affected by changes in atmospheric carbon dioxide (CO<sub>2</sub>) concentrations, temperature, precipitation, and nutrient levels. Successional forces lead to somewhat predictable **changes in forest stands** throughout the world. These changes can cause corresponding shifts in the dynamics of carbon uptake, storage, and release.

- Forest stands accumulate carbon as they progress through successional stages. Most studies show that the greatest rate of carbon uptake occurs during the stem exclusion stage, but mature stands also sequester and store significant quantities of carbon. This is even the case for old growth – particularly when such old forests represent significant portions of large areas such as the Amazon and Congo basins.
- Free Air Carbon dioxide Enrichment (FACE) experiments are suggesting that forest net primary productivity, and thus carbon uptake, usually increases with higher levels of atmospheric carbon dioxide, likely due to factors such as increased nitrogen use and water use efficiency and competitive advantages of shade tolerant species.
- Experiments dealing with drought and temperature change are providing evidence that water availability, especially soil moisture, may be the most important factor driving forest carbon dynamics.

#### *Caveats*

- Although we understand the stages of stand development, there is considerable unpredictability in the actual nature of species composition, stocking, and rates of development at each stage because of numerous positive and negative feedbacks that make precise understanding of future stand development difficult.
- Forest ecosystem experiments, such as FACE programs, have not been operating long enough to predict long term responses of forest ecosystems to increases in carbon dioxide. The expense and time constraints of field experiments force scientists to rely on multifactor models (the majority of which account for five or fewer variables) leading to results based on broad assumptions.

**Soil organic carbon** (SOC) stored and cycled under forests is a significant portion of the global total carbon stock, but remains poorly understood due to complex storage mechanisms and inaccessibility at depth.

- Alterations of soil carbon cycling by land use change or disturbance may persist for decades or centuries, confounding results of short-term field studies. Such differences must be characterized, and sequestration mechanisms elucidated, to inform realistic climate change policy directed at carbon management in existing native forests, plantations, and agroforestry systems, as well as reforestation and afforestation projects.

- Fine roots are the main source of carbon additions to soils, whether through root turnover or via exudates to associated mycorrhizal fungi and the rhizosphere.
- Bacterial and fungal, as well as overall faunal community composition, have significant effects (+/-) on soil carbon dynamics; fossil fuel burning, particulate deposition from forest fires, and wind erosion of agricultural soils are thought to affect microbial breakdown of organic matter and alter forest nutrient cycling.

#### *Caveat*

- The global nature of the carbon cycle requires a globally-distributed and coordinated research program, but thus far research has been largely limited to the developed world, the top 30 cm of the soil profile, temperate biomes, and agricultural soils. Forest soils in tropical moist regions are represented by only a handful of studies and even fewer have examined sequestration of mineral soil carbon at depth.

#### **CARBON BUDGETING AND MEASUREMENT**

- Quantifying carbon sources and sinks is a particular challenge in forested ecosystems due to the role played by biogeochemistry, climate, disturbance and land use, as well as the spatial and temporal heterogeneity of carbon sequestration across regions and forest types.
- While forests have the capacity to sequester significant amounts of carbon, the natural and anthropogenic processes driving carbon fluxes in forests are complex and difficult to measure. Nevertheless, accurate measurement of carbon stocks and flux in forests is one of the most important scientific bases for successful climate and carbon policy implementation. A measurement framework for monitoring carbon storage and emissions from forests should provide the core tool to qualify country and project level commitments under the United Nations Framework Convention on Climate Change, and to monitor the implementation of the Kyoto Protocol.
- Land use change is widely considered the most difficult component to quantify in the global carbon budget. Current consensus is that carbon emissions from land use change have remained fairly steady over the last few decades; however, there have been significant regional variations within this trend. Specifically, deforestation rates in the tropics, particularly in Asia, have grown substantially. In contrast, forests outside the tropics have been sequestering incremental carbon due to increased productivity (possibly because of CO<sub>2</sub> fertilization, although the evidence is not clear) and forest re-growth on lands that had been cleared for agriculture prior to industrialization.

- There are four categories of methods currently used to measure terrestrial carbon stocks and flows: i) the inventory method, based on biomass measurement data; ii) remote sensing techniques using satellite data; iii) eddy covariance method using CO<sub>2</sub> flux data from small experimental sites; and iv) the inverse method, using CO<sub>2</sub> concentration data and transport models. Each has advantages and disadvantages and varying degrees of accuracy and precision. No single method can meet the accuracy and resolution requirements of all users. A country, user or site will make a choice of method based on the specifics of the circumstances.
- Climate change is likely to generate both positive and negative feedbacks in forest carbon cycling. Positive feedbacks may include increased fire and tree mortality from drought stress, insect outbreaks, and disease. Negative feedbacks may include increased productivity from CO<sub>2</sub> enrichment. While the net result from positive and negative climate feedbacks is generally thought to be greater net carbon emissions from forests, the timing and extent of these net emissions are difficult to determine.
- Forest products are a minor, but growing component of the global carbon budget; nevertheless, harvested wood products can be long term reservoirs of carbon, particularly through substitution for more fossil carbon-intensive materials.
- Recycling postpones carbon emissions of even short-lived harvested wood products, and is especially effective when products are transformed multiple times within a tight recycling chain and finally converted into bioenergy.

### *Caveats*

- If a standardized verification system across projects, countries, and regions is ever to be attained, policymakers should be aware that there are different basic approaches to measuring forest carbon, which have advantages and disadvantages, and varying degrees of accuracy and precision.
- Land use change is widely considered the most difficult component to quantify in the global carbon budget. The underlying data is often incomplete and may not be comparable across countries or regions due to different definitions of forest cover and land uses. Deforestation rates in the tropics are particularly difficult to determine due to these factors as well as differences in the way land degradation, such as selective logging and fuelwood removals, are accounted for in national statistics.
- Knowledge of the amount of carbon stored within each pool and across forest types is limited. Even estimates using broad categories such as carbon in vegetation versus soils vary widely due to a lack of data or assumptions about where carbon is stored within the forest and at what rate carbon is sequestered or released.



- New processed wood products and paper manufacturing require large energy and heat inputs, making wood products and carbon a complex topic.
- Landfilling harvested wood products creates high levels of methane, and if capture systems for energy are not in place, then the potential of landfills to act as carbon sinks becomes very unlikely. Therefore, landfill gas capture systems must be required if this end-of-use pathway is to be promoted as a way to reduce carbon emissions.
- The substitution effects on greenhouse gas emissions of wood for other construction materials (e.g., steel and concrete) may be up to 11 times larger than the total amount of carbon sequestered in forest products annually. However, quantification of substitution effects relies on many assumptions about particular counterfactual scenarios, most importantly linkages between increased/decreased forest products consumption and total extent of forestland.

## **TROPICAL FORESTS**

Tropical forests contribute nearly half of the total terrestrial gross primary productivity and contain about 40% of the stored carbon in the terrestrial biosphere, with vegetation accounting for 58% and soil 41%. This ratio of vegetation carbon to soil carbon varies greatly by tropical forest type. About 8% of the total atmospheric carbon dioxide cycles through these forests annually. Vast areas of the world's large intact forests are in the tropics. Nevertheless, because of high rates of deforestation, tropical forests play a disproportionate role in contributing to terrestrial biome CO<sub>2</sub> emissions that both affect and mitigate climate.

- First and foremost, the primary risk to the carbon stored in tropical forests is deforestation, particularly converting forests to agriculture. Expanding crop and pasture lands have a profound effect on the global carbon cycle as tropical forests typically store 20-100 times more carbon per unit area than the agriculture that replaces them.
- In addition to the important role the remaining large intact forests play in the global carbon cycle, their protection from land conversion yields highly significant co-benefits. Evidence suggests that large, intact forests have significant cooling effects on both regional and global climates through the accumulation of clouds from forest evapo-transpiration, which also recycles water and contributes to the region's precipitation.
- Intact forests exist because of the geography of remoteness: low populations, lack of foreign investment, and lack of government presence have resulted in poor infrastructure development and the inability to integrate these regions into larger market structures.

- The significant drivers of deforestation (transportation infrastructure, agricultural commodity prices, national economic policies, agricultural technologies) are frequently context-specific and are affected by local political, socioeconomic, cultural, and biophysical factors. The roles of population growth and poverty in driving deforestation have often been overstated for certain regions (Africa may be an exception).
- The difference between the annual stand level growth (uptake: 2%) and mortality (release: 1.6%) of Amazonia is currently estimated to be 0.4%, which is just about enough carbon sequestered to compensate for the carbon emissions of deforestation in the region. This means that either a small decrease in growth or a small increase in mortality in mature forests could convert Amazonia from a sink to a source of carbon.
- CO<sub>2</sub> emitted from tropical soils is positively correlated with both temperature and soil moisture, suggesting that tropical rain forest oxisols are very sensitive to carbon loss with land use change.
- Old growth ever-wet and semi-evergreen forests are experiencing accelerated stand dynamics and their biomass is increasing, particularly in Amazonian and Central African forests, potentially in response to increased atmospheric CO<sub>2</sub>.
- Contrary to past assumptions, a significant portion of stored carbon exists below ground in tropical forests. Current estimates of root soil carbon in tropical forests could be underestimated by as much as 60%.
- If drought becomes more common in tropical ever-wet and semi-evergreen forests, as some climate models predict, the likelihood of human-induced fires escaping and impacting large portions of the landscape increases.
- Reduced impact logging (RIL) is an important practice to lessen carbon loss, but it is necessary to move beyond RIL to substantially increase carbon storage by developing more sophisticated, planned forest management schemes with silvicultural treatments that ensure regeneration establishment, post establishment release, and extended rotations of new stands.
- Land managers should not manage tropical forests only for timber production, but also to maximize and diversify the services and products they obtain from their forests. This approach will provide an increase in net present value and a possible solution to the problem of exploitation and land conversion.
- The largest potential source of carbon sequestration in the tropics is the development of second growth forests on old agricultural lands and agricultural plantation systems that have proven unsustainable. Every incentive should be provided to encourage this process. Many logged over and second growth forests are ideal candidates for rehabilitation through

enrichment planting of supplemental long-lived canopy trees for carbon sequestration.

- The overarching issues to be decided in developing an international policy to reduce emissions from tropical deforestation and forest degradation (REDD) include: the scope of the forestry activities to be covered; the scale of accounting for forestry activities and the baseline for measuring reference emissions levels; the type of financing to be provided for REDD activities; how to address fundamental issues of capacity and governance; and the consideration of co-benefits.

### *Caveats*

- Large intact forests of the tropics are increasingly at risk of deforestation attributable to governmental stimulus plans, road building programs, and subsidies for livestock production.
- A lack of governance, coupled with the presence of infrastructure, is often a precondition for widespread illegal operations that promote deforestation (e.g. logging, illicit drug trade). On the other hand, a lack of governance with no infrastructure inhibits illegal operations that promote deforestation.
- Tropical dry deciduous and montane forests are almost a complete unknown because so little research has been done on these forest types. While the majority of dry deciduous forests in the Americas and Asia have been cleared, there is still a significant amount remaining in Africa.
- Uncertainties in both the estimates of biomass and rates of deforestation contribute to a wide range of estimates of carbon emissions in the tropics. Only three studies have analyzed land surface-atmosphere interactions in tropical forest ecosystems. It is essential to understand how carbon is taken up by plants and the pathways of carbon release, and how increasing temperatures could affect these processes and the balance between them.
- More research is needed on how the application of silvicultural practices affects carbon uptake and storage in tropical forests at all levels. Some work has been done in the rainforest regions (ever-wet and semi-evergreen), but only in very specific places; almost none has been done in montane or seasonal (dry deciduous) forests.
- It is clear that REDD policies are only part of the solution to reduce deforestation and promote carbon sequestration. What is required is a combination of policies and market mechanisms that simultaneously promote sustainable economic growth and reduce poverty and economic inequalities, while protecting forests from further clearing for agriculture.

## TEMPERATE FORESTS

Twenty-five percent of the world's forests are in the temperate biome. They include a wide range of forest types, and the exact boundaries with boreal forests to the north and tropical forests to the south are not always clear. There is a great variety of species, soil types, and environmental conditions which lead to a diversity of factors affecting carbon storage and flux. Deforestation is not a major concern at the moment, and the biome is currently estimated to be a carbon sink of about 0.2 to 0.4 Pg C/year, with most of the sink occurring in North America and Europe.

- The future of the temperate forest biome as a carbon reservoir and atmospheric CO<sub>2</sub> sink rests mainly on its productivity and resilience in the face of disturbance. The small “sink” status of temperate forests could easily change to a “source” status if the balance between photosynthesis and respiration shifts even slightly.
- There is tremendous variability in carbon stocks between forest types and age classes; carbon stocks could easily be lost if disturbance or land use change shifts temperate forests to younger age classes or if climate change shifts the spatial extent of forest types. On the other hand, if temperate forests are managed for longer rotations, or more area in old growth reserves, then the carbon stock will increase.
- Temperate forests have been severely impacted by human use – throughout history, all but about 1% have been logged-over, converted to agriculture, intensively managed, grazed, or fragmented by sprawling development. Nevertheless, they have proven to be resilient – mostly second growth forests now cover about 40-50% of the original extent of the temperate forest biome.
- Soil carbon under temperate forests appears to be stable under most disturbances, such as logging, wind storms, and invasive species, but not with land use change. Huge losses can occur when converting forests to agriculture or development.
- Temperate forests are strongly seasonal, with a well-defined growing season that depends primarily on light (day length) and temperature. This is probably the most important determinant, along with late-season moisture, of temperate forest productivity and hence carbon sequestration.
- Natural disturbances, particularly windstorms, ice storms, insect outbreaks, and fire are significant determinants of temperate forest successional patterns. The frequency of stand-leveling windstorms (hurricanes, tornadoes) is expected to increase under a warmer climate in temperate moist broadleaf and coniferous forest regions, so that fewer stands would reach old-growth stages of development.

- If changing climate alters the frequency and intensity of fires, re-vegetation and patterns of carbon storage will likely be affected, particularly in interior coniferous forests.
- Storage of carbon in forests has played a major role in U.S. emission reduction efforts, particularly in the voluntary carbon markets. Considerable efforts have been underway to reduce emissions of greenhouse gases at the regional (Northeastern U.S.), state (California), municipal, corporate, and individual levels.

### *Caveats*

- Atmospheric pollution, primarily in the form of nitrogen oxides (NO<sub>x</sub>) emitted from burning fossil fuels, and ozone (O<sub>3</sub>) is a chronic stressor in temperate forest regions. Because most temperate forests are considered nitrogen-limited, nitrogen deposition may also act as a growth stimulant (fertilizer effect). Under current ambient levels, nitrogen deposition is most likely enhancing carbon sequestration; however, the evidence regarding long-term chronic nitrogen deposition effects on carbon sequestration is mixed.
- Data on mineral soil carbon stocks in temperate forests can only be considered approximations at this time as there is very little research on deep soil carbon (more than 100 cm).
- Global circulation models predict that increasing concentrations of atmospheric CO<sub>2</sub> will increase the severity and frequency of drought in regions where temperate forests are found. However, there is a great deal of uncertainty about how drought will affect carbon cycles.
- Although afforestation and reforestation projects are being considered under various global and national carbon policies, it is important to consider whether it is ecologically beneficial for the land to support trees. Afforestation or reforestation activities that require soil drainage or conversion of wetlands, as well as those that add stress to water-scarce areas, could create more public detriment than benefits.

### **BOREAL FORESTS**

As one of the largest and most intact biomes, the boreal forest occupies a prominent place in the global carbon budget. While it contains about 13% of global terrestrial biomass, its organic-rich soils hold 43% of the world's soil carbon. At present this forest biome acts as a weak sink for atmospheric carbon. However, the conditions that make this true are tenuous, and evidence of rapid climate change at northern latitudes has raised concern that the boreal forest could change to a net source if the ecophysiological processes facilitating carbon uptake are sufficiently disrupted.

- Increased fire frequency could greatly increase carbon release, especially if it increases the decomposition of “old” carbon from the soil pool by increasing soil temperatures, degrading permafrost, and enhancing the rate of heterotrophic respiration.
- While fire is recognized as the dominant natural disturbance type over much of the boreal forest, insect outbreaks (and “background” insect damage during non-outbreak years) are also critically important. In some forest types, insect outbreaks exert the primary influence on age class distribution.
- It appears that climatic warming is shortening the fire return interval in many boreal forests, and speeding up the life cycles of damaging insects. This could result in a large release of carbon, quickly turning the boreal forests from a sink to a source of carbon. Canadian forests in particular are poised to release massive amounts of carbon as the result of die-off from insect infestations.
- The question of whether moisture availability will decline with climatic warming will probably determine whether warming enhances the boreal carbon sink or turns it into a source.
- Lichens and bryophytes in lowland saturated sites contain upwards of 20% of the above ground carbon. These communities have important effects on how carbon is stored in boreal soils. Thick moss layers limit heat gain from the atmosphere, creating cold and wet conditions that promote the development of permafrost, with limited decomposition, thus are important for carbon storage.
- If all the carbon pools, inputs and outputs are considered together, it appears that clearcut stands in boreal forests are carbon sources for the first decade after harvest (thanks to transient increases in respiration), after which they switch to sinks.

### *Caveats*

- There is a tremendous amount of uncertainty in estimates of boreal carbon pools, because there have been so few studies in relation to the vast extent of the biome, and most have been done only in Canada and Fennoscandia.
- There is little quantifiable information about several important carbon pools, including fine root biomass and mycorrhizae, bryophyte and understory layers, and coarse woody debris and litter in Russia.
- Considering the importance of fire in boreal carbon dynamics, there is much that is not well understood, including extent, frequency, and intensity across the biome; and the interactions among fire intensity, nitrogen, and carbon.

## MANAGING TEMPERATE AND BOREAL FORESTS FOR CARBON

Increasing forest carbon stocks in temperate and boreal regions is a matter of making adjustments to existing forests vs. undergoing extensive reforestation/afforestation. Most boreal and temperate forests are second growth and land conversion is minimal when compared to other regions of the world. Therefore, providing additional carbon storage is a matter of refining silvicultural practices to take advantage of site nuances and enhancement potential.

- Many forest management activities result in net carbon release and thus cannot demonstrate carbon additionality. Mechanisms should be developed to credit managers who can reduce carbon loss, not simply increase carbon gain.
- Resiliency treatments (such as fuel reduction thinning and prescribed fire) result in lowered vegetative carbon storage, but they help produce forests that are significantly less susceptible to catastrophic disturbance (with accompanying drastic carbon release).
- Regeneration harvests significantly reduce the carbon stocks in vegetation and also cause a transient increase in soil respiration, although the annual rate of carbon uptake will be greater in the regenerating stand. Harvested areas often remain net carbon sources for 10-30 years, after which they return to sinks.
- Drainage of wetlands for increased tree production can result in either net carbon gain or loss, depending on how deep the drainage is.
- Studies have shown that many forest practices have a minimal impact on the soil carbon pool, which is the most difficult pool to measure. Thus, it may be possible that offsets involving certain forestry practices could go forward without strict quantification of this pool. This should be tempered by the fact that little is known about the effects of harvesting on deep soil carbon pools
- Managing stands for maximum sustained yield or financially optimum rotation can result in non-optimal carbon storage. Such rotations are often too short to allow the stand to attain maximum biomass. As such, it is often possible to increase carbon sequestration by extending rotations.

### *Caveat*

- If old forests *already exist*, however, it is almost never better to convert them to younger forests. Old-growth forests, especially in productive zones, often have very large pools of vegetative, bryophyte, and soil carbon in comparison to younger, managed forests.

## SUMMARY

Forests are critical to the global carbon budget, and every effort should be made to conserve intact forests, whether they are primary tropical and boreal forests, or second growth, temperate forests. All evidence points to the global forest estate being a weak sink for atmospheric CO<sub>2</sub>, as a result of a tenuous balance between the carbon sink from productivity in the temperate and boreal biomes and the net CO<sub>2</sub> emissions from the tropics due to large-scale deforestation. Changes in disturbance regimes (fire, storms, insect outbreaks, harvesting) in any of the major forested regions could easily tilt this balance one way or the other. And as these forests mature, their capacity to take up increasing levels of carbon commensurate with increases in CO<sub>2</sub> emissions will diminish. Future climate change effects on the forest carbon balance are difficult to predict: however, higher temperatures are likely to significantly influence the factors driving disturbance such as moisture, storms, and pest species ranges. Evidence of a “CO<sub>2</sub> fertilization effect” on forests is mixed, therefore it is difficult to predict whether or not continued increases in atmospheric CO<sub>2</sub> will counteract the negative influence of changes in disturbance frequency and intensity. Land use change, however, overwhelms all other factors, since continued deforestation in the tropics will most certainly push the “global forest” to being a net source of carbon emissions to the atmosphere instead of the sink it could be.



# Top 10 Recommendations for Preserving Carbon Stocks and Sinks in the World's Forests

1. **Keeping forests as forests** (i.e. preventing deforestation) is the most straightforward way of maintaining carbon stocks and promoting sequestration.
  - a. It is especially important to conserve intact primary forests.
  - b. Laws and economic policies that facilitate deforestation and forest degradation must be changed (for example, land tenure laws that promote deforestation or concession systems that allow poor harvesting practices and cause forest degradation).
2. **Reforestation on appropriate sites** is a viable means to enhance carbon sequestration.
  - a. Where NOT to plant: naturally treeless areas – montane grasslands, steppe, prairie, and tropical peatlands
  - b. In afforestation/reforestation projects, soil carbon must be included in carbon stock accounting.
3. Forests are dynamic systems. **In order to maintain resilient forests with lower risk of catastrophic carbon loss, it is sometimes necessary to undertake management practices that lower carbon stocks** (e.g. fuel reduction thinnings in fire prone forests).
4. **Setting a baseline (of carbon stock) against which to measure future gains** for carbon sequestration projects is an important policy choice, and will influence which “carbon positive” activities are implemented by landowners.
5. Forest carbon projects **must not damage ecosystem services** (water quality/yield, biodiversity, air quality).

6. When implementing forest carbon sequestration projects, efforts need to be made to **minimize shifting of deforestation to other areas (leakage)**.
  - a. Activity leakage: There is a risk that by delaying forest harvest in one place (through carbon sequestration projects), it will simply be shifted to other areas.
  - b. Market leakage: The desire to increase carbon sequestration in forests should not discourage wood use in favor of more fossil-carbon intensive products.
7. **U.S. climate policy should include international forests** (as offsets and/or through a fund).
8. In order for all countries to participate in a forest carbon regime, many will need **capacity building related to monitoring, forest management (e.g. zoning, operations and planning), and governance**.
9. **Equity: Forest dependent communities should be included in REDD policy decision-making and receive benefits from carbon projects**.
10. Market vs. Fund Mechanism: A **hybrid financing system** allows for a variety of forestry and climate change objectives to be met:
  - a. Markets can serve as a direct and consistent means for carbon offset credit values and transactions between suppliers and buyers over the long-term.
  - b. Funds can support activities like capacity building, pilot projects, and conservation that may not be intrinsically valued in a market framework.

Compiled by participants in the Yale School of Forestry & Environmental Studies graduate seminar *Managing Forests for Carbon Sequestration: Science, Business, and Policy*: Benjamin Blom, Jaime Carlson, Matthew Carroll, Ian Cummins, Cecilia Del Cid-Liccardi, Mark Evidente, Lauren Goers, Lisa Henke, Thomas Hodgman, Timothy Kramer, Janet Lawson, Eliot Logan-Hines, Brian Milakovsky, Jacob Munger, Caitlin O’Brady, and Ramon Olivas.

# Glossary

**Aboveground biomass**

Living vegetation above the soil, including stem, stump, branches, bark, seeds, and foliage.

**Additionality**

A criterion often applied to greenhouse gas (GHG) reduction projects, stipulating that project-based GHG reductions should only be quantified if the project activity would not have happened in the absence of the revenue from carbon credits and that only credits from projects that are “additional to” the business-as-usual scenario represent a net environmental benefit.

**Afforestation**

Planting of trees on historically non-forested land, e.g. native grasslands.

**Agroforestry system**

A mixed agricultural system that can combine planting of trees with agricultural commodities such as crops or grasses.

**Agrosilvopastoral system**

An agricultural system combining trees and livestock with agricultural crops and pasture.

**Albedo**

A surface’s reflectivity of the sun’s radiation. White surfaces, such as snow, cement/pavement or bare soil, have a high albedo, reflecting the sun’s radiation; dark surfaces, such as tree foliage or water bodies, have low albedo, absorbing more of the sun’s radiation.

**Allometry**

The study of the relationship between size and shape of organisms; in forestry, generally the relationship between tree diameter, height, crown size and biomass.

**Anoxic**

Soil conditions without oxygen.

**Annex I countries**

Parties to the United Nations Framework Convention on Climate Change (UNFCCC) that include the industrialized countries that were members of the OECD (Organisation for Economic Cooperation and Development) in 1992 as well as countries with economies in transition (the EIT Parties), including the Russian Federation, the Baltic States, and several Central and Eastern European States.

**Autotroph**

An organism which synthesizes organic materials from inorganic sources such as light (phototrophic) or chemical processes (chemotrophic); green plants and bacteria are autotrophs.

**Belowground biomass**

The living biomass of roots greater than 2 mm diameter.

**Biomass**

The total mass of living and /or dead organic matter found within a unit area usually measured as dry mass in grams, kilograms or tons per meter squared or per hectare.

**Bromeliad**

A diverse family of plants found chiefly in the tropical Americas that usually use the support of trees for their position in a forest canopy. Such plants are called epiphytic. Other bromeliads grow on the ground. Most have leaves arranged as rosettes. Bromeliads include the pineapple family, Spanish moss and various ornamentals.

**Carbon allowance**

Government-issued authorization to emit a certain amount of carbon into the atmosphere. In carbon markets, an allowance is commonly denominated as one metric ton of carbon dioxide, or carbon dioxide equivalent (CO<sub>2</sub>e).

**Carbon offset**

A financial instrument aimed at a reduction in greenhouse gas emissions. Carbon offsets are measured in metric tons of carbon dioxide-equivalent (CO<sub>2</sub>e) and are frequently generated by projects in sectors such as renewable energy and forestry. Offsets can be sold either in voluntary or compliance markets to an individual or company in order to compensate for greenhouse gas emissions or to comply with caps placed on emissions in certain sectors.

**Carbon sequestration**

The removal and storage of carbon from the atmosphere in carbon sinks (such as oceans, forests or soils) through physical or biological processes, such as photosynthesis.

**Carbon Tracker**

A system developed by the National Oceanic and Atmospheric Administration that calculates carbon dioxide uptake and release at the Earth's surface over time.

**Cation exchange capacity**

The capacity of a soil for ion exchange of cations (positively charged ion) between the soil and the soil solution and is used as a measure of fertility, nutrient retention capacity, and the capacity to protect groundwater from contamination. Plant nutrients such as calcium and potassium are cations, as are toxic metals such as aluminum.

**Clean Development Mechanism (CDM)**

A project-based mechanism defined in Article 12 of the Kyoto Protocol which allows a country with an emission-reduction or emission-limitation commitment under the Kyoto Protocol to implement emission-reduction projects in developing countries. Such projects can earn saleable certified emission reduction (CER) credits, each equivalent to one ton of CO<sub>2</sub>, which can be counted toward meeting Kyoto targets.

**Chronosequence**

A sequence of related soils or vegetation that differ from one another in certain properties primarily as a result of time as a soil-forming factor or succession, respectively.

**Coppice**

A traditional method of woodland management in which young tree stems are repeatedly cut down to near ground level so they will sprout into vigorous re-growth of young stems.

**Deadwood**

Non-living woody biomass either standing, lying on the ground (but not including litter).

**Deforestation**

Cutting down all the trees on a piece of land to convert it to another land use, or the long-term reduction of the tree canopy cover below a minimum 10 percent threshold.

**Developed land**

Urban and built-up areas.

**Disturbance**

Any event such as fire, wind, disease, insects, ice, flood, or landslide that disrupts the vegetation and abiotic environment in an area.

**DOC**

Dissolved organic carbon – see below.

**DOM**

Dissolved organic matter comprises carbon compounds in water solution, generally from decomposition of plant and animal tissues in soils.

**Ecological succession**

The relatively predictable change in the composition and/or structure of an ecological community, which may be initiated either by formation of new, unoccupied habitat (such as a severe landslide) or by some form of disturbance (such as fire, severe windthrow, logging) of an existing community.

**Ectomycorrhizal fungi**

A symbiotic association between a fungus and the roots of a plant that forms an important part of soil life and nutrient uptake in some forests.

**Eddy covariance**

A method of carbon measurement from forests that samples three-dimensional wind speed and CO<sub>2</sub> concentrations over a forest canopy at a high frequency and determines the CO<sub>2</sub> flux by the statistical relationship (covariance) of vertical wind velocity and CO<sub>2</sub> concentration.

**Epiphytes**

A plant that grows upon another plant (such as a tree) non-parasitically or sometimes upon some other object (such as a building or a telegraph wire), derives its moisture and nutrients from the air and rain and sometimes from debris accumulating around it, and is found in the temperate zone (such as mosses, liverworts, lichens and algae) and in the tropics.

**Extensive agriculture**

System of crop cultivation using small amounts of labor and capital in relation to area of land being farmed. The crop yield in extensive agriculture depends primarily on the natural fertility of the soil, terrain, climate, and the availability of water.

**Ex-ante accounting**

A method of accounting for emissions reductions in which money is given up-front for the guarantee that a given activity will be carried out and emissions reductions will occur in the future.

**Ex-post accounting**

A method of accounting for emissions reductions in which money is given for an emissions reductions activity after it has delivered its emission reduction.

**Fine root turnover**

The period of time for the fine roots of plants to form, function and then die.

**Floristics**

A sub domain of botany and biogeography that studies distribution and relationships of plant species over geographic areas.

**Forest**

Defined by the Food and Agriculture Organization as land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds *in situ*. It does not include land that is predominantly under agricultural or urban land use.

**Forest degradation**

Changes within the forest which negatively affect the structure or function of the stand or site, and thereby lower the capacity to supply products and/or services.

**Forest dynamics**

Describes the underlying physical and biological forces that shape and change a forest over time, or the continuous state of change that alters the composition and structure of a forest. Two basic elements of forest dynamics are forest succession and forest disturbance.

**Free Air Carbon Dioxide Enrichment (FACE)**

A method and infrastructure used to experimentally enrich the atmosphere enveloping portions of a terrestrial ecosystem with controlled amounts of carbon dioxide (and in some cases, other gases), without using chambers or walls.

**Fragmentation**

The transformation of a contiguous patch of forest into several smaller, disjointed patches surrounded by other land uses.

**Greenhouse gas**

Gas that traps heat in the atmosphere. The main greenhouse gases in the Earth's atmosphere are carbon dioxide, methane, nitrous oxide, and ozone.

**Gross primary productivity (GPP)**

The total amount of carbon compounds produced by photosynthesis of plants in an ecosystem in a given period of time.

**Heterotroph**

An organism capable of deriving energy for life processes only from the decomposition of organic compounds, and incapable of using inorganic compounds as sole sources of energy or for organic synthesis. Most animals are heterotrophic and rely on directly or indirectly (carnivores) eating most plants that are "autotrophic."

**Highest and Best Use (HBU)**

An appraisal and zoning concept that evaluates all the possible, permissible, and profitable uses of a property to determine the use that will provide the owner with the highest net return on investment in the property, consistent with existing neighboring land uses.

**Infiltration**

The process by which water on the ground surface enters the soil.

**Intensive agriculture**

An agricultural system with high productivity per unit area. Intensive agricultural systems also frequently have high input requirements per unit area, relying upon the use of mechanization, fertilizers, and agrochemicals.

**Kyoto Protocol**

A protocol to the United Nations Framework Convention on Climate Change (UNFCCC). It is an international environmental treaty negotiated in 1997 with the goal of stabilizing the concentration of greenhouse gases in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system.

**Land rent**

An economic term defined as the total net revenue or benefits received from a parcel of land.

**Land-use change**

The shift from one use of a land area to another, such as from forestry to agriculture.

**Leakage**

Term applied when activities that reduce greenhouse gas emissions in one place and time result in increases in emissions elsewhere or at a later date. For example, reduction of deforestation in one area of a country may lead to displacement of that deforestation to another region of the country.

**Liana**

Any of various long-stemmed, usually woody vines that are rooted in the soil at ground level and use trees, as well as other means of vertical support, to climb up to the forest canopy in order to get access to light; they are especially characteristic of tropical moist deciduous forests and rainforests.

**Lignin**

A complex chemical compound most commonly found in wood, and an integral part of the secondary cell walls of plants and some algae.



**Litter**

Forest carbon pool that includes the detritus, leaves, small dead biomass lying on the ground, and humus layers of the soil surface.

**Net primary productivity (NPP)**

The amount of carbon retained in an ecosystem (increase in biomass); it is equal to the difference between the amount of carbon produced through photosynthesis (GPP) and the amount of energy that is used for respiration (R).

**Non-Annex I countries**

Term referring to parties to the United Nations Framework Convention on Climate Change that are considered developing countries and were not required by the Kyoto Protocol to undertake national targets to quantify emissions reductions.

**Non-timber forest products**

Any commodity obtained from the forest that does not involve harvesting trees for wood products or pulp (paper products), such as game animals, nuts and seeds, berries, mushrooms, oils, foliage, medicinal plants, or fuelwood.

**Orographic precipitation**

Rain, snow, or other precipitation produced when moist air is lifted as it moves over a mountain range.

**Parcelization**

The breaking up of a land area under single ownership into multiple smaller parcels, usually for resale.

**Pasture**

Agricultural systems containing forage crops and used for grazing animals.

**Peatland (peat swamp forests)**

Tropical moist forests where waterlogged soils prevent dead leaves and wood from fully decomposing, which over time creates thick layers of acidic peat (organic matter).

**Permanence**

The longevity of a carbon pool and the stability of its carbon stocks within its environment.

**Photosynthesis**

The process by which a plant combines sunlight, water, and carbon dioxide to produce oxygen and sugar (stored energy and growth structure).

**Photosynthetically active radiation**

The spectral range (wave band) of solar radiation from 400 to 700 nanometers that photosynthetic organisms (e.g. plants) are able to use in the process of photosynthesis.

**Pioneer species**

Species which colonize previously bare or disturbed land, usually leading to ecological succession. Since uncolonized land may have thin, poor quality soils with few nutrients, pioneer species are often plants with adaptations such as long roots and root nodes containing nitrogen-fixing bacteria, and tend to grow well in open high-light environments.

**Plantation**

Forests planted as crops for the production of timber fruit, latex, oil or pulpwood. Many large industrial plantations are monocultures.

**Primary forest**

“Old” forests that have not been cleared by humans for a long period of time and have developed under natural ecological processes.

**Radiocarbon**

A radioactive isotope of carbon that is the most common for radiometric dating techniques.

**REDD+**

A climate mitigation policy being negotiated under the United Nations Framework Convention on Climate Change consisting of policy measures to create incentives for reduction of emissions from deforestation and forest degradation, conservation, sustainable management of forests, and enhancement of forest carbon stocks in developing countries.

**Reforestation**

Planting trees on land that was previously forested.

**Resiliency**

The capacity of a system to absorb disturbance and reorganize while undergoing change so as to retain essentially the same function, structure, and ecosystem services.

**Respiration**

The process by which animals and plants use up stored foods (mostly complex carbohydrates) by combustion with oxygen to produce energy for body maintenance.

**Rhizosphere**

The area immediately around plant roots, including the root itself that comprises intense microbial activity, where plants, microorganisms, other soil organisms, and soil structure and chemistry, interact in complex ways.

**Roundwood**

Harvested trees intended for use in products such as solid wood products, engineered wood products, and paper.

**Secondary forests**

Forests that have regenerated by natural processes following the clearance of primary forests by humans or a change in land use, for example, to agriculture, and then abandoned to revert back to forest.

**Silviculture**

The art and science of controlling the establishment, growth, composition, health, and quality of trees (woody plants) to meet diverse needs and values of the many landowners, societies, and cultures.

**Stand**

A group of trees of similar age-class, composition and site quality.

**Structural adjustment**

Term used to describe the policy changes implemented by the International Monetary Fund (IMF) and the World Bank in developing countries. These policy changes are conditions for getting new loans from the IMF or World Bank, or for obtaining lower interest rates on existing loans.

**Soil organic carbon (SOC)**

The carbon pool that includes all organic material in soil, but excluding the coarse roots of the belowground biomass pool.

**Soil microorganisms**

There are five major groups of soil microorganisms. Bacteria, fungi, actinomycetes, algae, and protozoa. Viruses form a small portion of soil microflora. They can be classified as autotrophs (utilize inorganic minerals) and heterotrophs (utilize organic matter).

**Thermokarst**

A land surface that forms as ice-rich permafrost thaws. It occurs extensively in Arctic areas, and on a smaller scale in mountainous areas such as the Himalayas and the Swiss Alps.

**Thinning**

The common term for the process of judiciously removing certain individual trees to improve the remaining quality and tree vigor in the plantation or forest; thinning can reduce the risk of a reversal of carbon sequestration due to fire, windthrow, insect infestations and disease.

**Throughfall**

The process by which precipitation has fallen through the vegetative (forest) canopy, including rain or fog that collects on leaves and branches.

## Yale Student Author Biosketches

**Benjamin Blom** MF '10 is a graduate of Colgate in political science. He is fluent in Spanish and Bahasa Indonesia. His concentration at Yale is in forest management and climate mitigation in tropical forests of Asia and Latin America.

**Jaime Carlson** MEM/MBA '09 is a graduate of Tufts in Biology. She has been involved with conservation and development in Panama. She is fluent in Spanish and had a concentration at Yale in forest finance and green investment. She is currently a Recovery Act Fellow at the U.S. Department of Energy.

**Matthew Carroll** MF '10 has degrees from Paul Smith's College and College of the Atlantic in Forestry and Human Ecology respectively. He has twelve years as a Wildland firefighter, both as a hotshot and smokejumper for the USFS. He has a concentration in forest management on public lands of the U.S. inland West in relation to climate change.

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## Editor Biosketches

**Mary L. Tyrrell** is the Executive Director of the Global Institute of Sustainable Forestry at the Yale School of Forestry & Environmental Studies. Her work focuses on land use change, forest fragmentation, sustainable forest management, and U.S. private lands, with a particular emphasis on review and synthesis of scientific research, and making scientific information available to forest managers, policy makers, and conservationists. She is the project manager of the Sustaining Family Forests Initiative, a national coalition focused on research and education about family forest owners in the United States. Ms. Tyrrell is a member of the Board of Advisors of the New England Forestry Foundation; the Board of Directors of the Hamden Land Conservation Trust; and Chair of the Environmental Concerns Committee at Saint Thomas More Chapel in New Haven.

**Mark S. Ashton** is the Morris K. Jessup Professor of Silviculture and Forest Ecology at the School of Forestry and Environmental Studies, Yale University. Professor Ashton conducts research on the biological and physical processes governing the regeneration of natural forests and on the creation of their agroforestry analogs. The results of his research have been applied to the development and testing of silvicultural techniques for restoration of degraded lands and for the management of natural forests for a variety of timber and nontimber products. Field sites include tropical forests in Sri Lanka and Panama, temperate forests in India and New England, and boreal forests in Saskatchewan, Canada. He has authored or edited over ten books and monographs and over 100 peer-review papers relating to forest regeneration and natural forest management.

**Deborah Spalding** is a founder and Managing Partner at Working Lands Investment Partners, LLC, which specializes in the investment and long-term stewardship of sustainably-managed working lands. She has worked in the financial industry for more than 17 years, serving in senior executive positions in the U.S. and overseas. Ms. Spalding is the Coordinator for Special Projects at the Yale School Forests, and serves on several boards, including the National Wildlife Federation, where she is a member of the Executive Committee, the Connecticut Forest & Park Association, and the Guilford Land Conservation Trust. She is a Trustee of the NWF Endowment and the Robert & Patricia Switzer Foundation, where she chairs the investment committee.

**Bradford Gentry** is the Director of the Center for Business and the Environment, as well as a Senior Lecturer and Research Scholar, at the Yale School of Forestry and Environmental Studies. Trained as a biologist and a lawyer, his work focuses on strengthening the links between private investment and improved environmental performance. He is also an advisor to GE, Baker & McKenzie, Suez Environnement, and the UN Climate Secretariat, as well as a member of Working Lands Investment Partners and Board Chair for the Cary Institute of Ecosystem Studies.

## **Global Institute of Sustainable Forestry**

Since its founding in 1900, the Yale School of Forestry & Environmental Studies has been in the forefront in developing a science-based approach to forest management and in training leaders to face their generation's challenges to sustaining forests. The School's Global Institute of Sustainable Forestry continues this tradition, in its mission to integrate, strengthen, and redirect the School's forestry research, education, and outreach to address the needs of the twenty-first century and a globalized environment. The Global Institute fosters leadership through dialogue and innovative programs, creates and tests new tools and methods, and conducts research to support sustainable forest management worldwide. The Global Institute works primarily through faculty-led programs and partnerships with other Yale centers and forestry institutions in the United States and abroad.

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The Center for Industrial Ecology at the Yale School of Forestry & Environmental Studies was established in September 1998 to provide an organizational focus for research in industrial ecology. The Center brings together Yale staff, students, visiting scholars, and practitioners to develop new knowledge at the forefront of the field. Research is carried out in collaboration with other segments of the Yale community, other academic institutions, and international partners. Faculty research interests include, among others, the theoretical basis of industrial ecology, the cycles of materials, technological change and the environment, eco-industrial urban development, industrial symbiosis, and product and producer policy issues.

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