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Forest Transitions and Carbon Fluxes

Global Scenarios and Policies

Edited by Matti Palo

December 1999

UNU World Institute for
Development Economics Research
(UNU/WIDER)

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This study has been prepared within the UNU/WIDER project on the Forest in the South and the North—Transition from Deforestation to Sustainable Forest Policies in Redressing Global Warming, directed by Professor Matti Palo with Dr Eustáquio J. Reis.

Professor Matti Palo is affiliated with the Finnish Forest Research Institute (METLA) in Helsinki.

UNU World Institute for Development Economics Research (UNU/WIDER) was established by the United Nations University as its first research and training centre and started work in Helsinki, Finland in 1985. The purpose of the Institute is to undertake applied research and policy analysis on structural changes affecting the developing and transitional economies, to provide a forum for the advocacy of policies leading to robust, equitable and environmentally sustainable growth, and to promote capacity strengthening and training in the field of economic and social policy making. Its work is carried out by staff researchers and visiting scholars in Helsinki and through networks of collaborating scholars and institutions around the world.

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FOREWORD

Climate change and the carbon cycle are in strong interaction with forests. Forestry operations impact on the climate, and vice versa. This study, which is based on the UNU/WIDER project on the Forest in the South and the North—Transition from Deforestation to Sustainable Forest Policies in Redressing Global Warming, discusses the scenarios and policies on how the deforestation of natural forests and the expansion of forest plantations influence the global carbon cycle through emissions and sequestration.

This publication constitutes a part of the research output from the above mentioned project. The editor of this report, Professor Matti Palo of the Finnish Forest Research Institute in Helsinki, was the External Director of the project, and Mr Eustáquio J. Reis in Rio de Janeiro was the External Project Coordinator. In addition to this publication, a policy-focused summary report of the findings entitled *Forests in Global Warming* was published by UNU/WIDER in 1998.

Launched during a three-day workshop in Nurmes (Finland) in June 1994, the project was divided into three sub-projects and has been fortunate in having the collaboration of other well-known research institutes. The case study on Brazil, under the guidance of Eustáquio J. Reis, was carried out with the assistance of the Institute of Applied Economic Research (IPEA/DIPES) in Rio de Janeiro. The Norwegian case study was conducted at the European Forest Institute in Joensuu, Finland, under the leadership of Birger Solberg. The study on tropical deforestation and the Chile sub-project were implemented at the Finnish Forest Research Institute (METLA) in Helsinki with Matti Palo as project director. In January 1996, a second workshop was organized jointly with the Center for International Forest Research (CIFOR) in Western Java, Indonesia. Focussing on theoretical and methodological aspects, the second workshop also reviewed the mid-term progress made in the three sub-projects. The concluding session of the project was held in October 1996 in Helsinki, Finland.

The theme of this publication has become most timely in the global political agenda in view of the follow-up of the Climate Change Convention ratified at UNCED in Rio de Janeiro in 1992 In the aftermath of the Kyoto, Buenos Aires and Bonn meetings, the role of forests as stocks, sinks and sources of carbon is still under considerable debate and controversy.

Giovanni Andrea Cornia
Director, UNU/WIDER
Helsinki, December 1999

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I also wish to thank all the contributors of the project for their devoted research and excellent cooperation. The authors come from six different countries and represent six disciplines, thus making the report truly international and multidisciplinary. Last but not least, we are grateful to the anonymous referees whose suggestions improved the content of the report.

Ms Liisa Roponen of UNU/WIDER was responsible for the language check, copy-editing and layout of the report and we wish to extend our sincere thanks to her. Finally, I would also like to thank Ms Barbara Fagerman of UNU/WIDER and Ms Margit Kuronen of METLA for their financial administration and secretarial support during the various phases of this project.

Matti Palo
Editor, Project Director
Helsinki, December 1999

ABSTRACT

Forests as stocks, sinks and sources of carbon have become a vital issue in global politics, along with the Kyoto Protocol of the Framework Convention on Climate Change. This publication, encompassing nine chapters by twelve authors from six countries, is the outcome of the respective research project conducted at UNU/WIDER.

As this study indicates, a very cost-efficient carbon conservation could be achieved by slowing down tropical deforestation and forest degradation. In this study, deforestation and carbon flux scenarios are modelled at the pantropical and continental levels, with special focus on Brazil, the country with the largest forest biomass of all countries of the world. Also afforestation and reforestation provide remarkable options as carbon sinks and stocks both in the south and north, as are illustrated by the case studies on Chile, Indonesia and Norway. The report also undertakes to analyse forest expansion and carbon fluxes in Europe and North America. The economics of tropical forestland use and global warming is also studied and, last but not least, international policy issues on carbon fluxes and forests in the south are reviewed. Replacing fossil fuels with woodfuels provides one policy option. Other options, including economic instruments such as taxes, offsets, and tradable permits, are also analysed.

Keywords: global forests, carbon emissions, carbon sequestration, carbon stocks, deforestation scenarios, carbon flux scenarios, forest expansion, international policy, north, south, tropics, tropical deforestation, plantation forests, Brazil, Chile, Indonesia, Norway

CHAPTER 1

GLOBAL SCENARIOS AND POLICIES ON FORESTS AND CARBON

Matti Palo and Birger Solberg[1]

1. INTRODUCTION

1.1 Purpose

This publication presents a part of the results from the UNU/WIDER research project 'The Forest in the South and the North—Transition from Deforestation to Sustainable Forest Policies in Redressing Global Warming', which was initiated in June 1994. The project focused on issues relevant for the follow-up of the process initiated by the UN Conference on Environment and Development (UNCED) in Rio de Janeiro in 1992. The specific research tasks of the project were:

- To provide causal explanations on the transition from deforestation towards sustainable forest management;
- To make scenarios of forest changes and carbon fluxes in the south and north in the short range (10–30 years) and the long range (50–100 years);
- To evaluate the effectiveness and cost efficiency of different forestry management options in sequestering CO_2 and in decelerating deforestation; and,
- To discuss the potential of tradable CO_2 permits in view of south-north cooperation on development and environment.

Nine papers commissioned within the project focused on the three last tasks, and are presented in this UNU/WIDER World Development Series report. The first task was addressed by another ten papers and these are being published by Kluwer Academic Publishers in their World Forests-series (Palo and Vanhanen, forthcoming).

This chapter gives an overview of the contents of the book, as well as our opinion on what are the major policy implications of the findings. In addition, the chapter presents the background for the research project.

1.2 Background

Policies for sustainable economic development require a more comprehensive understanding of global environmental change and its driving forces.

Man and forests have long lived together in harmony, with forests providing man with a wide range of goods and services. However, already for several millennia, clearing land for agriculture as well as setting deliberate fires to promote hunting and increase pasture have depleted forests as the global population has been growing.

As industrialization advanced, man began several centuries ago to subject the forests in the north to large-scale pressures. In the south, the same struggle appeared much later, during the second half of the twentieth century. Thus the large-scale and rapid deforestation, primarily in the tropics, has awakened the global community and forests became a part of the international political agenda from the 1980s.

At present, world forests extend to 3,400 million hectares (34 million km^2) which corresponds to 27 per cent of the world's total land area. Forests in the south and north are at different development stages. In the south, either in conjunction with dominating agriculture or early industrialization, deforestation of the natural forests is advancing at an annual rate of about 14 million hectares. In the post-industrial north, however, total forest area is increasing slightly, at about two million hectares annually (FAO 1997).

Global forests produce both marketed and non-marketed goods and services. The value of wood-based forest products is estimated at 2 per cent of the global GDP (FAO 1993). Forestry dominates in the south with fuelwood as the principal product, whereas in the north, forest industries generate more, relying primarily on various commercial products.

The total growing stock of global forests amounts to 384 billion m^3 and the corresponding biomass to 440 billion tons, of which (in dry mass equivalent) about half is composed of carbon. Furthermore, double that amount of carbon is sequestered in humus and soil as underground reservoirs. Thus, forest transitions have a major impact on global carbon fluxes. The average annual net release of carbon into the atmosphere from global forests due to deforestation is estimated at 0.9 ± 0.4 Pg (Pg = 10 in power of 15 or billion tons). This emission equals to about 16 per cent of the world's total carbon emissions from combusting fuels (IPCC 1996).

In addition to carbon, biodiversity within forests is another source of a global common good. It is approximated that some 13-14 million plants, animals and other living species exist on earth; of these, 50-90 per cent are assumed to be endemic only to tropical forests. As tropical deforestation continues, the risk of extinction of additional genes, species and ecosystems is increasing (UNEP 1995). Globalization of business, policy and culture appears to be one of the megatrends of this third millennium.

In the mid 1980s, UNCTAD influenced global forestry through the establishment of an international tropical timber agreement (ITTA), and its respective agency, the ITTO. In addition the tropical forestry action plan (TFAP) was created by the World Resources Institute, the World Bank, and FAO. The convention of International Trade in Endangered Species (CITES) was already ratified in 1975 but started to express concern over some tree species in the 1980s.

Particularly after the 1992 Rio de Janeiro UNCED-Conference, we see evidence of the introduction and strengthening of global forest policies. The Conference was a breakthrough with a number of documents concerning the forest sector, even though a global forestry convention never materialized despite time-consuming negotiations on the proposal. The most important UNCED forestry documents signed in Rio by the heads of states are:

- United Nations Framework Convention on Climate Change (UN 1992);
- Convention on Biological Diversity;
- Non-legally binding Statement on Principles for Global Consensus on the Management, Conservation and Sustainable Development of all types of Forests;
- Agenda 21, Chapter 11; and
- Rio Declaration.

The United Nations established a specific UN Commission of Sustainable Development (UNCSD) to follow-up on the implementation of the Rio agreements. UNCSD, in turn, set up the Intergovernmental Panel on Forests (IPF) which held four sessions during 1995-97, reporting the findings to its main body, UNCSD. UNCSD reviewed these findings in April 1997, and based on its recommendations, the United Nations' General Assembly Special Session (UNGASS) urged the international community in June of that year to establish an *ad hoc*, open-ended Intergovernmental Forum on Forests (IFF). Under the aegis of UNCSD, IFF was to promote and facilitate the implementation of the IPF's proposals for action. UNGASS also charged the Forum to identify possible elements of agreement on international arrangements and mechanisms, for example, a legally binding instrument to cover all types of forests in the south and north (UNCSD 1997), and to work towards achieving these goals. IFF will produce its policy recommendations in February 2000.

1.3 South-North challenges

During 1980-95, natural forests, despite the substantial global policy efforts described above, were depleted in the south by about 223 million hectares, while tree plantations increased forest area by 41 million hectares. Plantations are not, however, full substitutes for natural forests, particularly with regard to biodiversity and carbon reservoirs. But, plantations work well in producing fuelwood and pulpwood. China and India have by far the largest plantation forests and these have been primarily established as shelter belts against desertification and erosion, and for fuelwood production. Brazil, Indonesia and Chile have the largest commercial plantations, which have successfully been used for industrial purposes (FAO 1997).

Contrary to the south, forests in the north have been expanding throughout the second half of the twentithg century. From 1980 to 1995, forest expansion totalled 7 million hectares, but this is only a fraction of deforested areas of the south (FAO 1997). Traditionally the north has dominated the international trade of forest products with

Canada, Finland, and Sweden the biggest net exporters. Only a few countries in the south have been able to develop large-scale forest industries with comparative advantages in world markets. Malaysia and Indonesia provide examples of countries where sawmilling and plywood industries have been based so far primarily on natural forests. Brazil and Chile, to the contrary, have created viable pulp, paper and sawmilling industries based on plantation forests with exotic tree species (Palo and Mery 1996). However, in these four countries, deforestation of natural forests continues.

2. OVERVIEW OF THE PUBLICATION

2.1 Forest transitions, carbon fluxes and scenarios

Matti Palo, Erkki Lehto, and Raija-Riitta Enroth (Chapter 2) conduct scenario analyses on tropical deforestation and its consequences for carbon fluxes up to the years 2025 and 2050. The analysis comprises all natural tropical forests. The trend scenarios give the extreme limits assuming that deforestation will follow the development of the 1970s and 1980s. Practically all deforestation will take place in the period prior to 2025 according to the scenarios based on the three estimated regression models. The scenario models are constructed on the basis of causal modelling results to include population, income, forest data reliability, and ecological zones.

The regression scenarios project the total accumulated pantropical deforestation from 1990 to 2025 to be in the magnitude of 550-700 million hectares. This translates into an annual average rate of 16-20 million hectares of deforestation, compared to the 13-15 million hectares recently estimated by FAO. Forest area is expected to decrease up to the year 2025 in tropical America by 29-39, in tropical Africa by 37-41, and in tropical Asia by 31-36 per cent. A deceleration of deforestation is assumed to take place during the 2020s.

In typical tropical deforestation processes, forest is cleared for agricultural purposes by burning most tree-cover, after which the remaining dead biomass decays. During these processes, carbon dioxide is released into the atmosphere from both the biomass and soil. The carbon flux scenarios due to tropical deforestation are in the order of 70-100 gigatons of accumulated carbon emitted into the atmosphere, or 2.2-2.7 gigatons as an annual average over the period 1990-2025. These are higher than earlier scenarios (Chapter 2).

'Forest expansion and carbon fluxes in the north' are studied by Pekka E. Kauppi in Chapter 3. Direct and conclusive empirical evidence shows that the amount of vegetation biomass has increased in the forests of western Europe and the US, and has added to the sequestration of carbon dioxide from the atmosphere. According to indirect evidence, the amount of carbon in the soil has also increased in these northern forests, thus amplifying sequestration. However, the increase of carbon in the northern forests comprises only a fraction of the carbon decrease in the forests of the south. Moreover, the amount of carbon in wood products and landfills is on the rise, but in a lesser extent than the global carbon budget.

Chapter 4 by Eustáquio J. Reis concerns the modelling of 'carbon emissions scenarios for the Brazilian Amazon'. Deforestation of the Brazilian Amazon is recently estimated at 1.5 million hectares annually, and this generates about 2 per cent of the world's total carbon emissions.

The models designed by Reis consist of two main parts: the first describes the socio-economic interactions between infrastructure, population, land use, and deforestation, and the second describes the dynamics of carbon stock in the Amazon vegetation and soils related to different land uses. The socioeconomic interactions are estimated with econometric techniques applied to a panel of data on Amazon municipalities from 1975 to 1985.

Simulations of growth trajectories for the 1985-2010 period are made in the reference and counterfactual scenarios for the most important endogenous variables: urban and rural population, urban output, land uses, land yields, output in major rural activities, deforested areas, fallow areas, and carbon emissions.

In the reference scenario, deforestation in Legal Amazonia expands from 1985 to 2010 at a rate of 3.2 per cent per annum, compared to the rate of 9.5 per cent per annum observed in the 1980s. The slowdown is largely explained by the urbanization process and the increased turnover of fallow areas. Deforestation, as a per cent of the geographic area, increases from 8.7 per cent in 1985 to 19.8 per cent in 2010. Carbon emissions in the reference scenario decelerate significantly in the simulation period. Starting with 345 million tons per year in the 1985-90 period, they decline to 167 million tons per year by 2005-2010.

The estimated benefits per unit of carbon emissions are mch higher than the US$ 4/tC usually mentioned in the literature. This result holds even when benefits are measured exclusively by rural activity GDP. Finally, the growth of GDP per unit of carbon emitted is high. The main reasons are the increasing share of urban activities coupled to their higher productivity levels, the increase in agricultural productivity and the growing importance of biomass recomposition in secondary vegetation in fallow areas (Chapter 4).

'Plantation forests and carbon sequestration in Chile' with scenarios are investigated by Gerardo Mery and Markku Kanninen in Chapter 5. The Chilean forest sector has grown rapidly during the last 25 years, and currently plays an important economic role. This development has been based on the establishment of fast growing monocultures of radiata pine and eucalyptus plantations, which today extend close to 2 million hectares. The Chilean forestry is characterized by efficient wood production from forest plantations and the neglect of sustainable management of natural forests. During the last two decades, plantations have been established mostly on lands previously used for agriculture.

The role of plantation forests as a sink of atmospheric carbon dioxide is analysed by Mery and Kanninen, and a quantitative estimation is carried out to determine the current carbon pools and the annual fluxes. In the estimation of carbon sequestration, two essential factors are considered to influence volume increment: age distribution and latitudinal variation.

The amount of carbon sequestration of the Chilean forest plantations until the end of 1995 is estimated at 91.8 Tg and the annual flux for 1996 at 13.3 Tg yr^{-1}. It is stated that the annual carbon flux to forest plantations in 1996 exceeds annual carbon emissions from fossil fuels for that year. Five scenarios are constructed for simulating the carbon pool in radiata pine plantations between 1995 and 2025 under different management regimes. In these scenarios, the carbon pool increased by 47-98 per cent, depending on the management option and the annual planted area, thus emphasizing the need for a more balanced forest policy for promoting sustainable management of both natural forests and plantations. The policy for carbon management should be integrated into sustainable forest management as one of its basic criteria (Chapter 5).

2.2 Policy analyses

The substudy (Chapter 6) by Paula Horne covers 'the economics of tropical forest land use and global warming', primarily from a theoretical perspective. Tropical deforestation is of international concern, mainly because of the function of forests in safeguarding biodiversity and serving as carbon reservoirs. Less attention is paid to the needs of local people who might benefit from forest conversion, and this chapter focuses on extensive cultivation of land. It is one of the driving forces of deforestation, and characterizes the institutional and market failures that hamper the transition to more sustainable land use.

The optimization model presents a simplification of decision-making with regard to land use in tropical forests from the point of view of different interest groups. It shows how the optimization of land use under different property rights and different levels of internalization of externalities results in diverse preferences for the development of forest conversion. The possibilities to integrate global deforestation concerns to land-use practices which would benefit tropical nations and marginalized farmers are discussed in this chapter by Horne.

In Chapter 7, Hans Fredrik Hoen and Birger Solberg review the 'policy options in carbon sequestration via sustainable forest management in the north', with the purpose of presenting a detailed analysis of economic efficiency and potential of carbon sequestration of a boreal forest area. The analysis is based on data from the Norwegian national forest inventory for a region comprising of 695,000 hectares of productive forest.

Optimal forest management scenarios focusing on the first 30 years are calculated and production possibilities for commercial timber and carbon sequestration are investigated. Compared to earlier studies, this is based on a far more detailed representation of the forest data, a more realistic modelling of regeneration and natural mortality and a sensitivity analysis of the effects of an increase in natural mortality for old, dense stands. A major conclusion is that there are significant differences in carbon sequestration between a strategy of maximization of discounted timber profits and that of discounted net carbon dioxide sequestration (Chapter 7).

'Cost efficiency, environmental impacts and potential of carbon sequestration in Indonesia' are analysed by Birger Solberg and Boen Poernama in Chapter 8. By considering these three criteria jointly, the analysis shows that mixed plantations on grassland (*alang alang*) with *Paraserianthes falcataria* and *Acacia mangium* in short rotations (10 years) get high priority. Also, plantations of long-term rotations (60 years) of *Shorea spp.* give high carbon sequestration and economic returns. Both of these alternatives are the so-called 'no-regret' options, which means that they are profitable even if benefits of the carbon sequestration are discounted, e.g. the internal rates of return are respectively about 14 per cent per annum and 20 per cent per annum when alternative land-use values are assumed zero.

By comparison, monoculture plantations of *Paraserianthes falcataria* (10-year rotation) have about 40 per cent per annum higher carbon sequestration and 29 per cent per annum internal rate of return, but lower environmental score than the alternative mentioned above. The average annual carbon sequestration potential is about 55 million tons CO_2 for the 10-year rotation alternatives, and about 25 million tons CO_2 for the 60-year rotation alternatives, assuming that 10 per cent of the land available for the alternatives is utilized.

Although containing some uncertainty, the results indicate that the Indonesian forestry offers promising carbon sequestration possibilities. The challenge is to get international institutional arrangements organized so that these possibilities are optimally utilized (Chapter 8).

'International policy issues on carbon fluxes and forests in the south' are analysed by Carlos Young in Chapter 9. It is argued that global warming is a problem for everybody: the rich and the poor, the north and the south. Cooperation and participation endeavouring to control carbon emissions are vital, and solutions should not be restricted to a specific group of countries. But, equity principles cannot be overlooked either, and the north-south agenda for cooperation has to be based on fairness. Controlling the problem of greenhouse gases at the cost of increasing disparity between developed and developing countries is not a positive achievement.

Economic instruments have a crucial role in an efficient cooperation between developed and developing countries. The use of these economic instruments (carbon taxes, offsets and tradable permits) in the control of CO_2 emissions is analysed in Chapter 9. The preservation of tropical forests as carbon pools can be encouraged if industrialized countries provide economic incentives to 'import' these environmental services which today are given for free.

However, there are many issues that still need to be addressed. Allocation of emission rights, definition of compensation values, the possibility of free riding by non-cooperative countries, enforcement, settlement of disputes and other concrete problems are important institutional difficulties in the implementation of multilateral programmes. Even though there have been considerable advances in recent times, research in these topics remains important.

One specific research topic was developed more carefully—the proper evaluation of forest. To be effective, any compensation instrument should be based on a realistic valuation of the economic potential of the forests to be preserved. Nevertheless, some of the current proposals are based on the hypothesis that land prices reflect the true opportunity cost of land. One important finding of this study is that this procedure does not respect the principle of equity. The implementation of compensations based on actual land prices will induce the benefits accruing to the industrialized world to exceed the benefits of the countries that work effectively to control carbon emissions. This is shown in the exercises carried out for the Amazon. The risk of such a situation is that, instead of a true win-win solution, the wealth concentration problem is perpetuated to a global scale (Chapter 9).

3. POLICY PROPOSALS

3.1 Global cooperation

Globalization trends and a shift in forestry activities and forest industry investments from the north to the south bring new challenges to both hemispheres. Countries of the south and north are rather diverse, and are faced with different challenges. The diverging interests of the hemispheres with regard to global forest policy were first highlighted in the lengthy negotiations preceding the forestry principles of the Rio UNCED Conference in 1992.

At that time, the main theme was the control of tropical deforestation via a legally binding global forest convention. But as is pointed out in this study, instead of a global forest convention, the Rio meeting finally agreed upon voluntary forestry principles. Nevertheless, the consensus achieved on these principles has already had a strong effect on forestry paradigms and on practical applications of sustainable forest management, especially in the north but gradually also in the south.

The IPF and IFF processes have witnessed some shifting in the political arena, as is evidenced by the willingness on the part of previously reluctant countries to consider and to support an international convention. The resistance seen earlier to efforts to extend international forest policy is still strong. National sovereignty, and financial and trade-related issues still stand between the international community and a consensus on forests.

This research project has shown that there are a number of global forest policy-related options to promote the aims of both the Framework Convention on Climate Change and the Forestry Principles of Rio (UNCED). Moreover, the national sovereignty of individual nations to exploit their own natural resources and to implement national forest and environmental policies is also recognized. However, this report indicates that unless significant policy reforms are made, tropical deforestation continues and, according to our scenarios, such global commons as carbon reservoirs and sinks as well as the range of biodiversity are threatened. Even though most of the causes underlying deforestation rest in the hands of governments, the political will of national governments for large-scale deceleration of deforestation seems to be missing.

Efforts were made during the 1990s for a global forest convention, but so far without success. A few draft conventions, however, have been created (Humphreys 1996: Annexes A, D). A specific study was also prepared for the European Community on 'Options for strengthening the international legal regime for forests' (Glück *et al.* 1997). In addition to IFF's proposals which are expected by February 2000, the Intergovernmental Panel on Climate Change (IPCC) is working to clarify state of the art assessments regarding the impacts of land-use changes and forestry on the carbon cycle. Moreover, IPCC has an important global role with regard to the future and should, in our opinion, strive to support more research on the functions of the world forests in global warming.

3.2 Carbon balance control

The essential message of the two scientific assessments by IPCC (1990 and 1995) is that carbon dioxide remains the most important anthropogenic contributor to climate change. In IPCC's second assessment, forestry practices are mentioned as a means of promoting sustainable management of forests and, at the same time, of conserving and sequestering carbon. Among the practices recommended were various management approaches for conservation (emissions prevention), for carbon storage (short-range measures over the next 50 years or so) and for substitution (long-term measures).

It has been stated that the most cost-efficient carbon conservation would occur by reducing the rate of tropical deforestation and forest degradation. This can be done with storage management which means increasing the amount of carbon stored in vegetation, soil, and durable wood products. One complementary approach is to establish plantations. Substitution management, on the other hand, involves extending the use of forests for wood products and fuels. In the long range (more than 50 years or so), this would offer a more efficient strategy than terrestrial carbon storage, which in time reaches a saturation point. However, forestery regimes to conserve and increase carbon storage offer an effective mitigatory option during a transition period that can take many decades to stabilize atmospheric concentration of carbon.

Solberg (1997) shows the potentially high economic value of increased carbon sequestration of boreal forests. Equally important, he points to an argument which is often forgotten in this debate: that, given the uncertainties regarding the impact of global warming, the increased forest stocks of carbon provide flexibility. If man's activities are ultimately proven to cause global climate change, increased stocks of carbon in wood biomass (as trees and/or forest industry products) will have a long-range positive effect in lowering the quantities of atmospheric CO_2. This provides extra time to introduce new energy technologies or other measures to decrease the emission of greenhouse gases. In addition, the increased stocks of roundwood will most likely lower the price of wood, thereby adding demand for bioenergy and wood products, and thus contributing to a society based on renewable energy resources instead of fossil fuels.

But in transforming the present rapid rate of deforestation to sustainable forestry management, a controversy of global versus national interests exists. The benefits of carbon sequestration and biodiversity sustainability are primarily global, but the costs are national and local. Such countries as Guyana, Surinam, Papua New Guinea and Gabon which are still almost entirely under forest cover, may have specific land-use policies to decrease their forest areas for conversion to some other land uses. If the global community wants to prevent the implementation of such national policies, effective compensation programmes need to be created.

Some international bodies, for example FAO and the World Bank, could be motivated to organize policy workshops where scenario-making researchers would meet policy makers. Various scenarios, such as the ones introduced in this publication, could be discussed and evaluated by policy makers and new scenarios with alternative assumptions could be produced. Such workshops could be useful learning processes for all parties and improve both future research and policies.

3.3 Plantation forests

Mery and Kanninen (in this volume) describe how the development path followed by Chile has led to a two-faced forest sector: on the one side, dynamic development of the plantation-based forest industries is evident, while on the other, the country is hampered by its backward perception of natural forests and its less-than-efficient industry based on these forests. Whether this strategy constitutes a sustainable approach to development is a matter of debate. At least on the part of natural forest management, this development in recent years clearly is not sustainable. But unfortunately a number of countries are following a similar path. This issue, having global relevance, should be studied more.

The Chilean scenarios of radiata pine show that plantation forestry, where the main objective is to produce material for forest industries, has a positive effect on carbon balance. Under the 'silvicultural optimum' and 'conservative management' scenarios, in which relatively long rotations are used, the carbon pool in 2015 is approximately twice the 1995 level. Even under the 'aggressive' forest management scenarios with shorter rotations and extraction of raw material in greater quantities, the carbon pool increases by approximately 50 per cent. This case illustrates the 'base-line' problem in carbon accounting, an issue which also needs more research.

3.4 Sustainable forest management

The forest sector offers major opportunities for sustainable development. The first priority should be to adopt a more balanced approach in which the potential of plantation forestry and rational management and utilization of natural forests could be combined and complemented. Carbon policy to promote the mitigation of the build-up of atmospheric carbon dioxide needs to be integrated in forest sector strategy for sustainable development.

This integration is central because it is unrealistic to expect forests to be managed solely for carbon sequestration purposes. On the contrary, forests must also be managed—albeit sustainably—for their traditional economic rewards. The transition towards sustainable forestry is expected to contribute not only to increasing the carbon sink effects, but also to halting significant net additions of carbon into the atmosphere caused by deforestation and other non-sustainable forestry practices.

The results of this study illustrate that tropical forestry offers a number of interesting carbon sequestration options that are environmentally acceptable and that, in addition to their carbon offsetting benefits, also produce high economic returns. One main challenge is to set up the institutional arrangements at the international level so that these options are optimally implemented, to the benefit of all main interested groups involved. We recommend that these 'no-regret' forestry options be prioritized for implementation in both tropical and non-tropical countries.

Joint implementation of these options by the north and south produces an international win-win solution. Tradable permits and other financial instruments are examples of potentially effective policy measures available for global use in the control of carbon dioxide. The follow-up of the international conventions of climate change implementation in Kyoto, Buenos Aires and Bonn has not specified and adequately validated the various policy instruments available. For the next meeting, however, more research findings and policy proposals will be accessible.

NOTES

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CHAPTER 2

SCENARIOS ON TROPICAL DEFORESTATION AND CARBON FLUXES

*Matti Palo, Erkki Lehto and Raija-Riitta Enroth**

1. INTRODUCTION

1.1 The problem

The atmospheric concentrations of greenhouse gases, with carbon dioxide (CO_2) and methane (CH_4) as principal components, have been increasing steadily over the last century or so with the likely consequence of changes in global climate through the so-called greenhouse effect. Increases in these gases are the direct result of combusting fossil fuels, certain industrial processes, and interventions in the global carbon cycle such as deforestation or other changes in land cover.

Tropical forests have been proven to be relatively large net carbon sources. Carbon fluxes in 1990 from tropical deforestation are estimated to be 1.65±0.4 GT/a (1 GT = billion tons = 1Pg = 10^{15}g). This is equivalent to almost 30 per cent of the annual emissions resulting from the use of fossil fuels (Brown 1996). There are many reasons to assume that this estimate may be even larger because of uncertainties in determining the rates of deforestation and forest degradation, and the fate of deforested land, which subsequently results in the carbon pools of forests being eliminated (Brown *et al.* 1993).

Uncertainty in the global assessment of the impacts of deforestation for CO_2 emissions and biodiversity loss, for example, is partially caused by the lack of comprehensive data on the rate of deforestation itself. Tropical deforestation is a diverse phenomenon. Causes and consequences are interlinked, and it is difficult to assess the relative effect of the various agents on deforestation. However, since the 1980s more research has been focused on the causes of deforestation as well as on the impacts of the process (e.g. Brown and Pearce 1994; Lambin 1994; Palo and Mery 1996; Kaimowitz and Angelsen 1998).

1.2 Purpose of this study

The purpose of this study is to make scenario analyses on tropical deforestation and its consequences to carbon fluxes. The time horizon of our scenario models in the short range will be to the year 2025 and to 2050 in the long range. The research problem is divided into three closely interrelated tasks as follows:

i) estimating trend models and multiple regression models on deforestation separately for tropical Africa, Asia and Latin America;

ii) producing scenarios on tropical deforestation, as based on trends and multiple regression models; and

iii) with the aid of these scenarios, producing scenarios on carbon fluxes as a consequence of tropical deforestation.

We have restricted our analysis to cover natural forests, which are defined as ecosystems with a minimum of 10 per cent crown cover of trees and/or bamboo on a particular land area. Natural forests are generally associated with indigenous wild flora, fauna, and natural soil conditions, and they are not subject to agricultural practices (FAO 1993 and 1995).

Forest plantations constitute about 2.5 per cent of all tropical natural forest areas (FAO 1993) and are not included in the analysis, because they are not full substitutes for natural forests and because their transition process follows different causal mechanisms. Also excluded from this study are trees outside the forest and in plots smaller than 100 hectares as well as other wooded land.

We limit the definition 'deforestation' to those changes in land cover which deplete the crown cover of natural forests to less than 10 per cent of forest land. Accordingly, on the other hand, deforestation is not applied to forest degradation, which reduces the growing tree stock, alters the distribution of species, or may degrade the soil of a forest for which the definition of 'natural forest' would still apply. Typically, forest degradation transforms closed forest into open forest (FAO 1993).

Forest area is used in the deforestation modelling instead of tree biomass or volume because of greater reliability. Later in the carbon flux scenarios, we have applied the same average biomass and carbon contents per hectare as in some earlier studies in order to maintain a basis for comparison. Thus, forest degradation has also been incorporated in the study (Section 4.3).

Our study includes 90 tropical countries (FAO 1993) of which 40 are located in Africa, 17 in Asia (and Pacific), and 33 countries in Latin America (and the Caribbean). In 1990, they comprised 1,756 million hectares of tropical natural forests or 98.0 per cent of the total forest area in all the world's 115 tropical countries. Our empirical data are based on the FORIS database of the Tropical 1990 Forest Resources Assessment (FAO 1993) while the 'Global Synthesis' of two years later (FAO 1995a) included another 25 small tropical countries with a total additional natural forest area of 36 million hectares.

2. PREVIOUS STUDIES

We found diverging results on the role of population pressure on deforestation. On the one hand, some researchers say that no convincing evidence has been found which would indicate that population pressure increases deforestation (Brown and Pearce 1994; IPF 1996). On the other hand, certain studies support the theory that population and deforestation are related (Reis and Guzman 1994; Palo and Lehto 1996; Palo and Vanhanen, forthcoming) and population pressure is frequently cited as a cause of deforestation. Population growth, by increasing the demand for food and other agricultural products, promotes the conversion of forest land to other uses. Similarly, population growth also increases the demand for forest products and various infrastructure services that hasten deforestation.

Changes in farming systems are an important mediating variable between demographic changes and deforestation or forest degradation. The agrarian change can be categorized as agricultural extensification, agricultural intensification, and agricultural involution (e.g. Lambin 1994). According to agricultural extensification models, increased demand can easily be satisfied by expanding the cultivated area. This is achieved by extending cultivation into virgin areas or by migrating into unsettled locations. These occur essentially in regions with a low population density and a high rate of population growth. Under these conditions, agricultural expansion into open access forest ecosystems results in deforestation.

Theoretical and empirical studies suggest that in extensive, low-input subsistence farming systems, added demand in consumption leads to an increase in the frequency of cultivation and a reduction of the fallow period. This directly causes a larger percentage of the area to be under forest fallow. Eventually with higher population densities, the fallow cycle becomes too short to allow natural reforestation to take place. See Horne (Chapter 6) for the impact of weak tenure and property rights in this extensification process.

Once all open access land has been utilized either for cultivation or fallow, or access has been denied, additional food demand caused by the population growth can be satisfied domestically only by increasing the intensity of agriculture. Despite a high rate of population growth, the autonomous adjustment mechanisms of technological innovation and agricultural intensification may be slow to develop—or may not develop at all—which leads to an overexploitation of soil resources.

It is also possible that fallow periods are reduced without an accompanying intensification because of the limited knowledge of the population, risk-averting attitudes, lack of credits or certain other reasons. In this case, overexploitation of the soil and declining productivity can lead to rising mortality or forced migration. Eventually, the environment is destroyed through excess utilization and involution, and the third process of deforestation in relation to agrarian change, becomes evident.

The link between population pressure and deforestation is a complex and indirect issue that involves interacting variables and processes. Demographic pressure can both create the need to exploit forests, but it also provides abundant labour to support the intensification of cultivation under strong and clear tenure conditions, but these are not common in the tropics. Thus, the influence of population pressure is only one element of a broader socioeconomic system affecting deforestation (e.g. Palo 1990; Lambin 1994).

While population pressure contributes to deforestation, economic growth and modern technology can modify its effects. For instance, take the case of two countries: both with rapid population growth rates and significant forest resources, but different levels of per capita income. In all likelihood, deforestation will develop at a slower pace in the higher-income country with clear and strong tenure conditions, because people, as income increases, switch to energy sources other than fuelwood and adapt modern agricultural techniques, thereby reducing pressure for clearing forests for agricultural purposes (Palo 1990; Cropper and Griffits 1994).

Income and its growth can have varying effects on deforestation: initially, higher incomes generate greater demand for agricultural and forest products, thus stimulating deforestation; next, better incomes provide resources for large, capital-intensive projects and agricultural subsidies that are often associated with deforestation; and finally, higher incomes and growing economies can lead to better employment opportunities, thus reducing the incentive of poor families to clear forest land in isolated areas. In environmental economics, this phenomenon is called the environmental Kuznets curve or inverted u-shape response of increasing income on the deterioration of the environment (Palo and Lehto 1996; Palo and Vanhanen, forthcoming).

Accessibility of forests is an important economic factor in deforestation: road construction and easier access to floating in forest areas increase deforestation (e.g. Lambin 1994; Reis and Guzman 1994; Panayotou 1994). Demand for tropical hardwoods, on the other hand, does not directly cause the type of deforestation, as implied by complete forest removal. Industrial logging, compounded with population pressure, has an indirect effect because the construction of roads facilitates access to forest clearing by marginal farmers under open access conditions. Accessibility has been approximated also by the distance from a metropolis. Furthermore, better soil quality leads to greater deforestation, while the poorer health environment often prevalent in rainforests has a tendency to slow down deforestation. Moist ecological zones as a proxy to inaccessibility have received empirical evidence lately (Palo and Lehto 1996).

Only a few studies on scenarios of pantropical deforestation and its consequences on carbon fluxes are available. Matsuoka *et al.* (1994) have estimated the carbon dioxide flux arising from tropical deforestation. Assuming a dominant role for population pressure in deforestation, they construct models based solely on the relationship between deforestation and population growth, and apply deforestation data from the FAO and UNEP 1980 assessment. With three scenarios for population change, their study estimates carbon dioxide flux during the 120-year period from 1980 to 2100 in three regions (Latin America, Africa and Asia).

Trexler and Haugen (1995) have assessed both the deforestation rates by gathering their own empirical data and the factors assumed to influence future deforestation in 52 selected tropical countries. These parameters include population growth, urbanization, agricultural trends, land-tenure policies, development of infrastructure, expansion of cash crops, government and land-use plans, extension programmes, public perceptions of forest resources, energy needs, laws and regulations. Instead of applying an objective modelling, Trexler and Haugen preferred to use a Delphi-type technique of inquiry addressed to experts in each country. They also estimated the change in carbon emissions associated with each land-use change.

Jepma (1995) developed a global intersectional policy simulation model based on system dynamics to analyse possible future processes of tropical forests. The simulations of forest areas were based on FAO's Production Yearbook which, according to Palo and Mery (1996) cannot produce as valid and reliable data on deforestation as the 1990 FAO assessment (1993).

Nilsson and Schopfhauser (1995) estimated the quantities of carbon which could be sequestered by a global forestation programme. Using many secondary sources, they estimated the amount of land likely to be available, feasible annual planting rates and rotation lengths for different countries and regions. For information on the tropics, Nilsson and Schopfhauser also relied on the above-mentioned studies (Trexler and Haugen 1995; Jepma 1995).

Accordingly, it would seem that earlier studies on pantropical deforestation and carbon flux scenarios, have been based on other data sources than the latest FAO assessment (1993) which forms the foundation for the empirical data of the scenario analyses given here.

3. MODEL SPECIFICATION AND DATA

3.1 Model specification

The model specification here is based on a previous causal modelling research by Palo and Lehto (1996) and Palo and Vanhanen (forthcoming). First, we experimented successfully with five different variations of dependent variables. Thus becoming familiar with the functioning of the different function forms and transformations of the independent variables, it convinced us of the validity of the expected signs in the model specification. The identification of the model is explained in Palo and Vanhanen (forthcoming)

Given the interdependence between the various factors of deforestation described above (Section 2), we assume that deforestation is primarily a function of forest land demand for other uses, primarily for agriculture. Also, the demand for forest products, especially for industrial logs and fuelwood, plays a role in this process. Accordingly, we assume that changes in both population and income affect deforestation. However, we also assume that ecological conditions and economic accessibility of forests are determinants of deforestation.

Because of the paucity of reliable data on roads, the economic accessibility of forests is measured according to ecological zones (wet or dry). The wettest zone is assumed to be the least accessible because of increased closed forests and biomass characteristics as well as limited infrastructure and deteriorating health conditions. The dry ecological zone, on the contrary, is assumed to be the most accessible due to diametrically opposite conditions (Palo 1987).

In addition, it is hypothesized that the quality of information affects deforestation. Better information on forest resources can induce great usage of forests and subsequently more deforestation. Also the general level of statistical data may measure the same phenomenon and have the same kind of effect. Our assumption here is based on the hypothesis that imperfect information increases the risk to the economic agents of deforestation. However, the quality of information may be context-bound so that better data on the scale and pace of deforestation at a later stage of the process may start to limit deforestation by creating counterbalancing forces (Palo and Lehto 1996; Palo and Vanhanen, forthcoming).

When causal models are transformed into scenario models, it is important to specify the rather simple models which consist of lagged variables and/or variables for which predictions are already available. Approximated lagged data forms for constructing trend models are also available for forest areas. Variables for population and economic growth projections exist at the national level, or if unavailable, it is plausible, based on past trends, to estimate the alternative future scenarios for them.

The dependent variable of the specified models here is forest cover, which is specified both as logarithmic and logistic functions (Palo and Vanhanen, forthcoming). The former is defined as forest area per land area (per cent), while the latter is measured as forest area per nonforest area (per cent) which describes deforestation as a process that tends toward saturation within a given geographic area or as a decreasing s-function. In most countries, there usually are inaccessible forest areas and/or in a later phase, some policy instruments are taken to decelerate deforestation (Palo 1987; Scotti forthcoming).

3.2 Constructing scenarios

It should be kept in mind that while the purpose of this study is to construct alternative scenarios, these are not predictions but are intended to merely provide an alternative image of a possible future and thus a conditional prediction to the question: What will happen if? For example, if population and GNP per capita were to change as assumed, then deforestation will follow a certain scenario. In order to map the future, a few alternative scenarios are created to describe not only the end points, but also the pathway to reach them.

We first produced scenarios based on the hypothesis of 'business as usual' according to which no major policy shift or radical changes in the causal system will affect deforestation rates and subsequent carbon releases. Next, we assumed future changes in population growth scenarios based on an authoritative UN (1995, 1996 and 1998) source, and GNP/capita (FORIS-database) based on national trends for 1981-90. Consequently, our deforestation scenarios are based on the most recent empirical data on forest areas, trends, and modelling of the factors, which affect future forest area changes and thus deforestation.

One major aim in our scenario modelling is to simulate possible future paths of deforestation in order to provide policy makers with effective tools in their efforts to advert the realization of the scenarios projected in this chapter.

3.3 Empirical data

The lack of reliable and valid data is the most difficult problem in modelling deforestation. Reliable time-series data on forest cover or deforestation are not available (Palo and Mery 1996). Also, the socioeconomic data of the developing countries may be inaccurate (Lambin 1994). Due to data limitations, many studies use cross-sectional data and forest cover as the dependent variable.

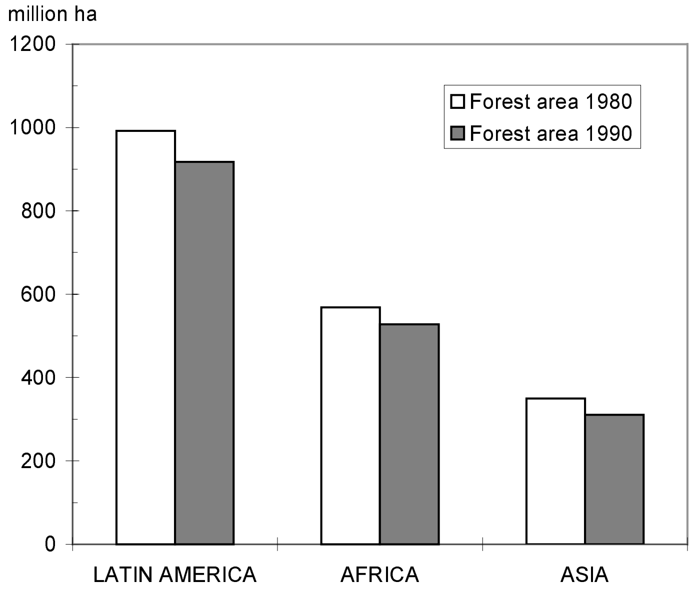
The forest cover (percentage or absolute) existing in a particular area is the function of two factors: amount of the original forest cover and the forest cover lost since the start of the deforestation process. In other words, the forest cover existing today is the result of natural factors and a history that may cover six thousand years of deforesting activity by man, as in some parts of China (Zhang, forthcoming), or a couple of months. On the other hand, Palo (1987) and Reis and Guzman (1994) argue that the countries or municipalities used in these analyses deviate from each other with regard to the stage of their development and can represent a long time-dimension of decades or centuries. This indicates that a time-dimension is necessary and has been included in our study.

Deforested area is perhaps the most valid dependent variable, but measurement errors are more common in this variable than in the variable of forest cover, because deforestation can be gauged only as an estimate of change which, to be fully analysed, would require two inventories to be conducted at different intervals with similar methods and degrees of accuracy. In a previous causal modelling (Palo and Lehto 1996; Palo and Vanhanen, forthcoming) which forms the basis of this scenario design, we were able to model the forest cover, the deforestation rate and three other relevant alternatives as dependent variables and found quite similar explanatory variables.

The primary data source for this study is the Tropical Forest Resource 1990 Assessment (FAO 1993) which provides data on forest area and its reliability, land area, ecological zones, GNP and demographic information for 90 tropical countries. Respective data for the modelling is drawn from the FORIS-database, primarily from the subnational level of 578 geographical units as well as partially from the national level (Palo and Vanhanen, forthcoming).

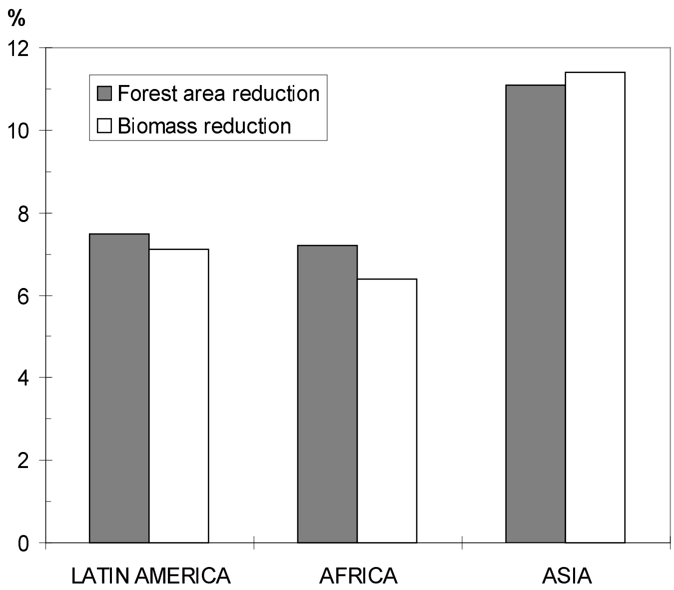
Deforestation in the tropics has advanced from 1981 to 1990 at an average annual rate of 15 million hectares and on each continent according to the scale of total forest areas (Figure 2.1). Loss of forest biomass has progressed approximately at a similar pace (Figure 2.2), with some variation by continent.

FIGURE 2.1
NATURAL FOREST AREA BY CONTINENTS



Data source: FAO (1995a).

FIGURE 2.2
REDUCTION IN NATURAL FOREST AREA AND TOTAL ABOVE-GROUND BIOMASS
IN THE TROPICS, 1980-90 (%)



Data source: FAO (1995a).

The UN (1995, 1996 and 1998) provides estimates and projections for the total global population. The projection horizon was extended to the year 2050 in order to present a demographic history and projection for an entire century (1950-2050), and three fertility variants and one fertility scenario of population projections were prepared for the period 1990-2050: the high-, medium-, and low-fertility variants; and the constant-fertility scenario. In our scenarios, we use the medium-fertility variant.

The world population was estimated at 2.5 billion people in 1950 and 5.6 billion in 1994. The medium variant scenarios project 8.2 and 9.8 billion people for 2025 and 2050, respectively (UN 1995). By 2100, world population may total 10.3 billion people but estimates range from 3.9 to 22.7 billion (Lutz 1996: 500).[1] Most of the anticipated increase will take place in the developing world and these prospects cause concern with regard to tropical deforestation. By 2050, the population of Africa will be about three times the current level, and population is projected to slightly less than double in Asia and Latin America, although relative growth rates will continue to decline.

Past national trends over the period 1981-90 are used to estimate the future development of GNP. The annual GNP/capita growth rates by continent range between 3.1 per cent for Asia and -1.1 per cent for Africa. The figures are rather pessimistic but we consider these more realistic than other available projections to the year 2050 according to continent only (Nakicenovic and Jefferson 1995). Low growth rate variants of GDP are estimated at 4.0 per cent after 2010 for all continents and high growth rate variants between 4.0–5.0 per cent. In any case, should there be need to apply different assumptions on GNP, population and forest area reliability scenarios, it is possible to recalculate the scenarios.

4. ESTIMATION AND RESULTS

4.1 Trend scenarios on deforestation

As the first approach, we estimated trend scenarios based on the assumption that the decrease of forest area can be proportional, linear, accelerating proportional or accelerating linear (Table 2.1). With this method, we endeavoured to map all possible future scenarios inherent in the internal empirical forest cover data.

The four scenarios differ, depending on the different assumptions of the nature of decrease. Results were computed for both the continental level and subnational level (Table 2.2). The decrease at the continental level is usually greater. Proportional decrease gives the smallest reduction in all continents. Accelerating linear decrease produces the largest reduction in Africa, Latin America and in the tropics as a whole. In Asia, the reduction is largest when linear decrease is assumed.

TABLE 2.1
DESCRIPTION OF TREND SCENARIOS 1a-4b

1 PROPORTIONAL TREND

Assumption: Annual average deforestation rate % remains the same as between 1981-90.

(a) Continental decrease

$FA_t = FA_{90} * e^{[(t-1990)/10]*Ln(FA_{90}/FA_{80})}$

where FA_t = Forest area in year t
FA_{90} = Forest area in year 1990
FA_{80} = Forest area in year 1980.

(b) As sum of subnational units

$FA_t = \Sigma FA_{90} * e^{[(t-1990)/10]*Ln(FA_{90}/FA_{80})}$.

2 LINEAR TREND

Assumption: Annual average deforested area remains the same as between 1981-90.

(a) $FA_t = FA_{90} - [(t-1990)/10]*(FA_{80} - FA_{90})$.

(b) $FA_t = \Sigma Max\{FA_{90} - [(t-1990)/10]*(FA_{80} - FA_{90}), 0\}$.

3 ACCELERATING PROPORTIONAL TREND

Assumption: Change in the proportional forest area change remains the same as between 1971-80 and 1981-90.

(a) $FA_t = FA(\text{by proportional trend 1a.}) * e^{([(t-1990)/10]*[((t-1990)/10)+1]/2)*Ln[(FA_{90}/FA_{80})/(FA_{80}/FA_{70})]}$

$= FA_{90} * e^{[(t-1990)/10]*Ln(FA_{90}/FA_{80}) + ([(t-1990)/10]*[((t-1990)/10)+1]/2)*Ln[(FA_{90}/FA_{80})/(FA_{80}/FA_{70})]}$

(b) $FA_t = \Sigma Min\{FA(\text{by proportional trend 1b.}) * e^{([(t-1990)/10]*[((t-1990)/10)+1]/2)*Ln[(FA_{90}/FA_{80})/(FA_{80}/FA_{70})]}, FA_{90}\}$

4 ACCELERATING LINEAR TREND

Assumption: Change in the linear forest area change remains the same as between 1971-80 and 1981-90

(a) $FA_t = FA(\text{by linear trend 2a.}) - ([(t-1990)/10]*[((t-1990)/10)+1]/2)*[(FA_{80}-FA_{90})-(FA_{70}-FA_{80})$

$= FA_{90} - [(t-1990)/10]*(FA_{80} - FA_{90}) - ([(t-1990)/10]*[((t-1990)/10)+1]/2)*[(FA_{80}-FA_{90})-(FA_{70}-FA_{80})]$

(b) $FA_t = \Sigma Min\{Max\{FA(\text{by linear trend 2b.}) - ([(t-1990)/10]*[((t-1990)/10)+1]/2) * [(FA_{80}-FA_{90})-(FA_{70}-FA_{80})], 0\}, FA_{90}\}$

According to our trend scenario results (Table 2.2 below), total tropical forest area could drop from the 1990 level of 1,756 million hectares to 802-1,170 million hectares by 2050. The relative average annual decrease (per cent) for the period 1990-2050 is the fastest in Asia, then Latin America and finally Africa. It is interesting to note that the maximum decreasing rates of deforestation according to continents run parallel to the GNP per capita development of each continent. Average annual growth rates for GNP per capita have been +2.0/+3.1 per cent in Asia, +1.8/-0.2 per cent in Latin America and 0/-1.1 per cent in Africa as measured trends during 1961-92 and 1981-90.

4.2 Regression scenarios on deforestation

4.2.1 Model estimation

We discovered the statistically best regression models for constructing deforestation scenarios by continents by applying stepwise regression analysis with subnational and national data (Table 2.3). The regression equations are estimated using logarithmic variables and ordinary least squares. All coefficients produced the expected signs, and the coefficients of wet ecological zone areas and national land areas were statistically significant in all models. We assumed the increase in deforestation to parallel increasing incomes at low levels. After a certain income level, increasing incomes will start to reduce deforestation (Palo and Vanhanen, forthcoming). In Africa and Asia, which have low income levels, the coefficients of GNP are, as expected, negative while in Latin America where the income level is high, the respective coefficients are positive.

The coefficients of population, GNP and reliability of forest data are mostly statistically significant in the models for Latin America and Africa. In Asia, however, the coefficients of population and GNP are not significant at the 10 per cent probability level. The risks, however, are not very high and range between 23 and 42 per cent. The reason for Asia's nonsignificance may be the fact that the number of observations for Asia is much smaller than for the other two continents examined. For example, in Asia there are 78 subnational units and 17 countries, and the corresponding numbers in Africa are 329 and 40, respectively. Population and GNP are also variables, which we modelled only at the national level for scenario construction.

However, our models 5-6 for Asia have higher adjusted R-squares (45-43 per cent) than comparable models 1-3 for Latin America and Africa (28-39 per cent). Asian model 5 also has the smallest standard error among the six models for the three continents (Table 2.3). All models have relatively strong multicollinearity (bottom row). Distribution of residuals may need some further analysis (cf. Palo and Vanhanen, forthcoming).

TABLE 2.2

TREND SCENARIOS FOR LATIN AMERICA, AFRICA, ASIA, AND TOTAL TROPICS TO THE YEARS 2025 AND 2050

	Scenario assumptions			Remaining forest area in:						Reduction from 1990								
				millions ha			% of land area			Total				Annual				
	Kinds of decreases									millions ha		%		millions ha		%		
				1990	2025	2050	1990	2025	2050	2025	2050	2025	2050	2025	2050	2025	2050	
LATIN AMERICA	Proportional	0.8%/a (1981-90)	1A	918	700	576	56	42	35	218	342	24	37	6.2	5.7	0.8	0.8	
			1B	918	732	642	56	44	39	186	276	20	30	5.3	4.6	0.6	0.6	Min.
	Linear	7.4 m. ha/a (1981-90)	2A	918	656	474	56	40	29	262	444	28	48	7.4	7.4	0.9	1.1	
			2B	918	678	548	56	41	33	240	370	26	40	6.9	6.2	0.9	0.9	
	Accelerating proportional	0.11%/a/10a (1971-90)	3A	918	643	460	56	39	28	275	458	30	50	7.9	7.6	1.0	1.1	
			3B	918	661	513	56	40	31	257	405	28	44	7.3	6.8	0.9	1.0	
	Accelerating linear	0.54 m. ha/a/10a (1971-90)	4A	918	616	360	56	37	22	302	558	33	61	8.6	9.3	1.1	1.5	Max
			4B	918	617	437	56	37	26	301	481	33	52	8.6	8.0	1.1	1.2	
AFRICA	Proportional	0.7%/a (1981-90)	1A	528	406	337	24	18	15	122	191	23	36	3.5	3.2	0.7	0.7	
			1B	528	409	347	24	18	16	119	181	22	34	3.4	3.0	0.7	0.7	Min
	Linear	4.1 m. ha/a (1981-90)	2A	528	384	282	24	17	13	144	246	27	47	4.1	4.1	0.9	1.0	
			2B	528	381	287	24	17	13	147	241	28	46	4.2	4.0	0.9	1.0	
	Accelerating proportional	0.05%/a/10a (1971-90)	3A	528	390	302	24	17	14	138	226	26	43	3.9	3.8	0.9	0.9	
			3B	528	389	312	24	17	14	139	216	26	41	4.0	3.6	0.9	0.9	
	Accelerating linear	-0.02 m. ha/a/10a (1971-90)	4A	528	386	286	24	17	13	142	242	27	46	4.1	4.0	0.9	1.0	
			4B	528	374	278	24	17	12	154	250	29	47	4.4	4.2	1.0	1.1	Max
ASIA	Proportional	1.2/%/a (1981-90)	1A	311	205	153	35	23	17	106	158	34	51	3.0	2.6	1.2	1.2	
			1B	311	220	181	35	25	20	91	130	29	42	2.6	2.2	1.0	0.9	Min
	Linear	3.9 m. ha/a (1981-90)	2A	311	174	76	35	19	9	137	235	44	75	3.9	3.9	1.6	2.3	Max
			2B	311	188	140	35	21	16	123	171	40	55	3.5	2.9	1.4	1.3	
	Accelerating proportional	0.02%/a/10a (1971-90)	3A	311	202	146	35	23	16	109	165	35	53	3.1	2.8	1.2	1.2	
			3B	311	209	167	35	23	19	102	144	33	46	2.9	2.4	1.1	1.0	
	Accelerating linear	-0.38 m. ha/a/10a (1971-90)	4A	311	204	156	35	23	18	107	155	34	50	3.1	2.6	1.2	1.1	
			4B	311	208	174	35	23	19	103	137	33	44	2.9	2.3	1.1	1.0	
TOTAL TROPICS	Proportional	0.8%/a (1981-90)	1A	1756	1308	1060	37	27	22	448	696	26	40	12.8	11.6	0.8	0.8	
			1B	1756	1362	1170	37	29	24	394	586	22	33	11.3	9.8	0.7	0.7	Min
	Linear	15.4 m. ha/a (1981-90)	2A	1756	1217	832	37	25	17	539	924	31	53	15.4	15.4	1.0	1.2	
			2B	1756	1247	975	37	26	20	509	781	29	44	14.5	13.0	1.0	1.0	
	Accelerating proportional	0.07%/a/10a (1971-90)	3A	1756	1236	911	37	26	19	520	845	30	48	14.9	14.1	1.0	1.1	
			3B	1756	1259	992	37	26	21	497	764	28	44	14.2	12.7	0.9	0.9	
	Accelerating linear	0.14 m. ha/a/10a (1971-90)	4A	1756	1206	802	37	25	17	550	954	31	54	15.7	15.9	1.1	1.3	Max
			4B	1756	1199	889	37	25	19	557	867	32	49	15.9	14.5	1.1	1.1	

Note: A - continental level; B - sum of subnational units

TABLE 2.3
SCENARIO REGRESSION MODELS BY SUBNATIONAL UNITS
OF TROPICAL LATIN AMERICA, AFRICA, AND ASIA

Region / Dependent variable	Latin America		Africa		Asia	
	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6
	Forest area (% of land area)	Forest area (% of nonforest area)	Forest area (% of land area)	Forest area (% of nonforest area)	Forest area (% of land area)	Forest area (% of nonforest area)
Independent variables	Coefficient (standard error in parenthesis)					
Intercept	2.74 ***	-2.52 ***	3.03 ***	x	2.30 ***	-2.52 **
	(0.39)	(0.73)	(0.54)		(0.65)	(1.05)
National population $_{t-10}$ (total)	-0.71 ***	-1.34 ***	-0.12	-0.31 **	-0.20	-0.32
	(0.20)	(0.37)	(0.09)	(0.13)	(0.17)	(0.27)
Gross national product $_{t-10}$ (total)	0.30 **	0.54 **	-0.20 **	-0.24 *	-0.11	-0.16
	(0.15)	(0.27)	(0.08)	(0.12)	(0.12)	(0.20)
Wet area ecological zone (ha)	0.15 ***	0.34 ***	0.07 ***	0.12 ***	0.13 ***	0.20 ***
	(0.02)	(0.04)	(0.02)	(0.04)	(0.03)	(0.04)
Dry area ecological zone (ha)	x	x	-0.07 ***	-0.10 ***	x	x
			(0.02)	(0.02)		
Reliability of forest data high (0.1)	-0.62 ***	-1.22 ***	-0.37 **	-0.68 **	x	x
	(0.14)	(0.27)	(0.18)	(0.28)		
Reliability of forest data low (0.1)	x	x	-0.53 ***	-0.75 ***	x	x
			(0.11)	(0.17)		
Population data from 1960 (0.1)	x	x	-0.32 ***	-0.60 ***	x	x
			(0.11)	(0.16)		
National land area (ha)	0.36 ***	0.77 ***	0.30 ***	0.40 ***	0.31 ***	0.50 ***
	(0.08)	(0.16)	(0.06)	(0.08)	(0.11)	(0.17)
Countries	33	33	40	40	17	17
Subnational units	171	171	329	329	78	78
Observations	167	167	305	305	75	75
Adjusted R square	0.30	0.39	0.28	0.61	0.45	0.43
R square	0.32	0.41	0.30	0.62	0.48	0.46
Standard error	0.86	1.60	0.83	1.31	0.65	1.05
F-statistic	15.3	22.0	15.6	59.3	16.3	14.7
Significance of F	0.00	0.00	0.00	0.00	0.00	0.00
Maximum pairwise Pearson correlation r (between independent variables in parenthesis)	0.87 (1.2)	0.87 (1.2)	0.76 (1.2)	0.76 (1.2)	0.87 (1.2)	0.87 (1.2)

Notes:
*** = significance level under 1%
** = significance level under 5%
* = significance level under 10%
x = statistically nonsignificant variable excluded in order to minimize standard error.

Models estimated from FORIS database (FAO 1993), forest area as dependent variable by subnational units of tropical Latin America, Africa and Asia (all variables in natural logatirthms; OLS estimation).

4.2.2 *Deforestation scenarios*

Next we introduce the deforestation scenarios based on the regression models (Table 2.4). By the year 2050, the total pantropical forest area is estimated to be between 1,070-1,200 million hectares. This scenario range matches the upper sector of the trend fork (Table 2.2, Figure 2.4). The essential difference between the trend scenarios and regression model scenarios is that deforestation in the latter scenarios is estimated to occur during the 1990-2025 period, which means a more rapid process of early deforestation, which we regard as the realistic alternative. In fact, the whole scenario range lies just below our trend fork by 2025 (Figure 2.4).

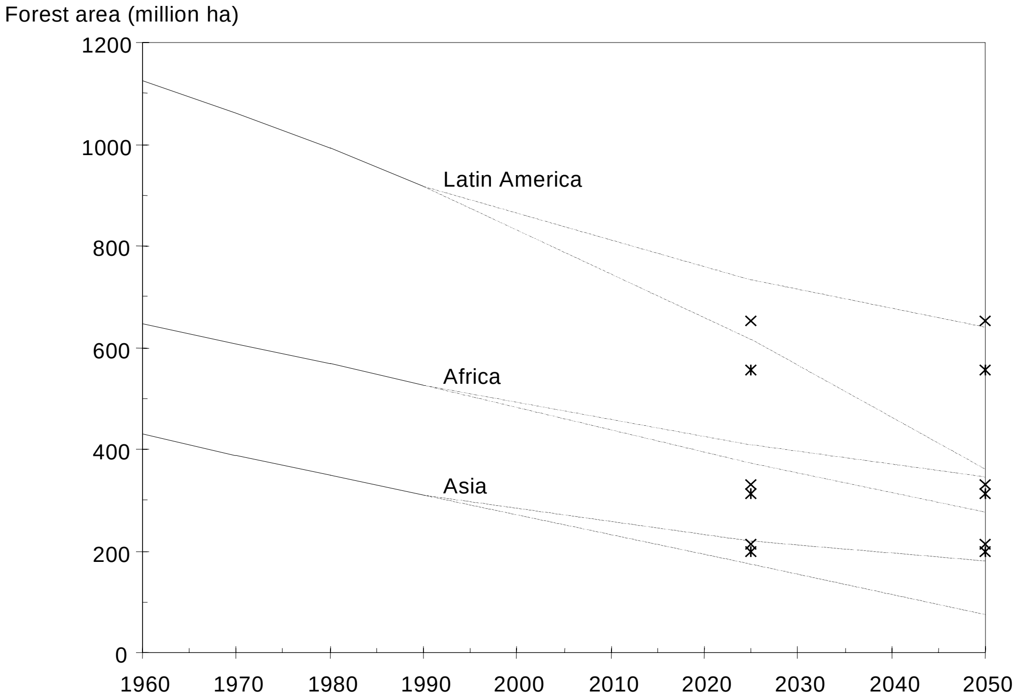
TABLE 2.4
SCENARIO MODEL ASSUMPTIONS AND RESULTS FOR LATIN AMERICA, AFRICA, AND ASIA

	Latin America		Africa		Asia		Total tropics	
	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Sum of models 1, 4, & 5	Sum of models 2, 3, & 6
Population growth rate (annual %)								
1990-2025	1.4	1.4	2.6	2.6	1.4	1.4	1.7	1.7
2025-2050	0.7	0.7	1.6	1.6	0.7	0.7	1.0	1.0
GNP/capita growth rate (annual %)								
1990-2050	-0.2	-0.2	-1.1	-1.1	3.1	3.1	0.9	0.9
Reliability								
2025	A	A	C	C	D	D		
2050	B	B	B	B	D	D		
Forest area remaining, million ha								
1990	918	918	528	528	311	311	1756	1756
2025	556	654	330	313	199	214	1068	1198
2050	556	654	330	313	199	214	1068	1198
Forest area remaining, % of land area								
1990	56	56	24	24	35	35	37	37
2025	34	40	15	14	22	24	22	25
2050	34	40	15	14	22	24	22	25
Total reduction 1990-2025								
million ha	362	264	198	214	112	97	688	559
%	39	29	37	41	36	31	39	32
Annual reduction over 1990-2025								
million ha	10.3	7.6	5.7	6.2	3.2	2.8	19.7	16.0
%	1.4	1.0	1.3	1.5	1.3	1.1	1.4	1.1

Note:
A High dummy=0.5 if High dummy=0 in 1990
B Dummies excluded
C High dummy=0.5 if High dummy=0 in 1990; Low dummy=0.5 if Low dummy=1 in 199(
D No dummies

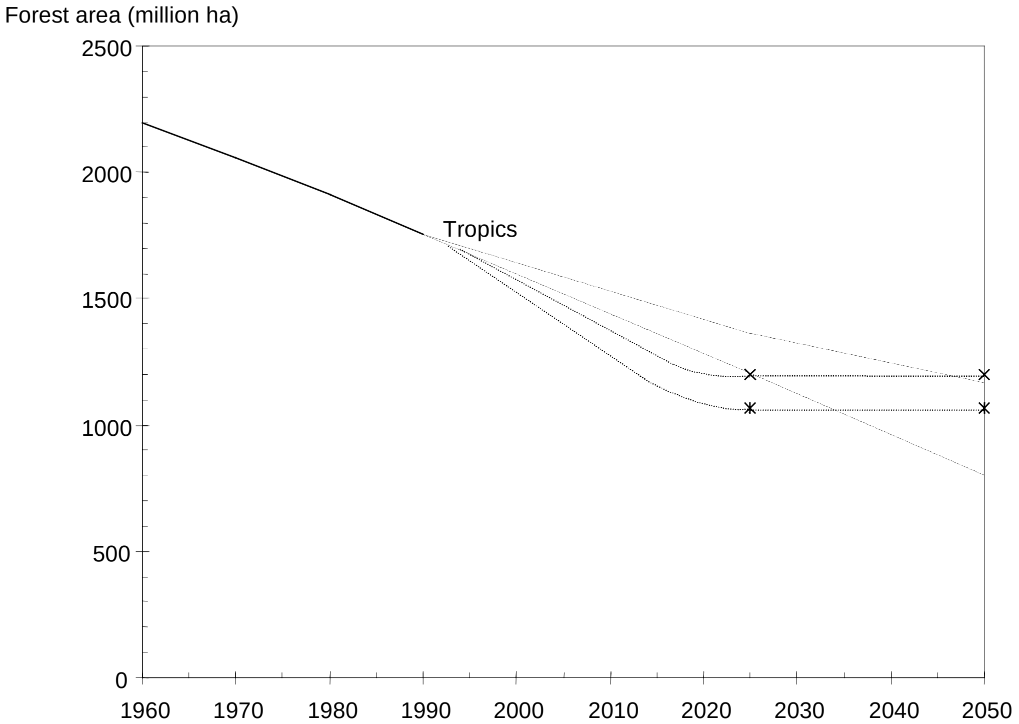
The continental deforestation scenarios produced by the two approaches are also compared (Figures 2.3-2.4). A faster rate of deforestation to the year 2025, based on regression modelling scenarios as compared to the trend scenario fork, will develop in Africa and partly in Latin America. By 2050, the regression scenarios in Asia appear even slightly above the respective trend scenario fork whereas in Africa and Latin America, the match of the two types of scenarios is fairly good.

FIGURE 2.3
NATURAL FOREST AREA SCENARIOS FOR TROPICAL LATIN AMERICA, AFRICA, AND ASIA



Note: The cut line indicates low and high trend scenario, the asterisk and the cross low and high model-based scenario.

FIGURE 2.4
NATURAL FOREST AREA SCENARIOS FOR TOTAL TROPICS



Note: The cut line indicates low and high trend scenario, the asterisk and the cross with a dotted line low and high model-based scenario.

4.3 Scenarios on carbon fluxes

4.3.1 Assessing carbon fluxes

Human-induced net releases of carbon from forests to the atmosphere through changes in forest cover depend on the quantities of carbon stored within the vegetation and the soil, rate and form of deforestation, forest degradation, and the intensity of soil disturbance. The amount of vegetation carbon in a hectare of natural forest can vary significantly, according to temperature, precipitation, soil regimes, land form, forest management, and other human or natural intervention. Thus, carbon estimates per hectare can deviate considerably even within a given country or district. Using approximations found in the literature, Trexler and Haugen (1995) have converted hectare estimates of biomass carbon densities for three tropical climatic zones. Given as tons per hectare, these are: 35-75 for dry, 75-125 for moist, and 125-225 for wet climatic types.

In this study, the estimates for carbon stored in vegetation per unit of forest area are based on the carbon pool and flux analyses conducted by Dixon *et al.* (1994), who classify the world's forests into three latitudinal belts: low-, mid-, and high-latitude forests that are located between 0 to 25, 25 to 50, and 50 to 75 latitudes, respectively. Nations or regions are grouped by latitude on the basis of the approximate geographic location of their forests. The estimated area-weighted carbon densities for forest vegetation and soil densities vary considerably within the continent (Table 2.5). As Dixon *et al.* (1994) point out, uncertainties exist for all forest groups, but major inconsistencies are in the low-latitude forests. Spatial variation in vegetation carbon density estimates may be up to 90 per cent of mean values. Although estimates of biomass carbon densities have been improved, these are in need of constant refinement to reflect the continuous forest degradation.

TABLE 2.5
ESTIMATED AREA-WEIGHTED CARBON DENSITIES IN FOREST VEGETATION AND SOILS

	Vegetation		Soil		Total (vegetation and soil)
	Carbon density (Mg/ha)	Reduction %	Carbon density (Mg/ha)	Reduction %	Carbon flux (Mg/ha)
Africa	99	70	120	30	105.3
Asia	132-174	70	139	35	155.75
Latin America	130	80	120	40	152

Source: Dixon *et al.* (1994); Brown *et al.* (1993); Houghton *et al.* (1987); FAO (1996)

Forest degradation often advances faster in tropical rainforests than deforestation. For example, the primary forests of the Malaysia Peninsular were reduced in area by 12 per cent, but lost about 35 per cent of their biomass carbon density during 1972-82 as a result of harvesting large-diameter trees. These estimates are based on two subsequent national forest inventories (Dixon *et al.* 1994). Also Trexler and Haugen (1995) point out that information is sketchy on the magnitude and patterns of degraded tropical forests which are not yet fully deforested, and the impact of these on carbon flows.

There are also wide variations in estimates of the organic-matter content of tropical forest soils. However in general, the range of guesstimates on soil carbon content is not as wide as for the estimates for living biomass (Nilsson and Schopfhauser 1995).

The amount of carbon flux during a deforestation process is determined by the type of land-use conversion and the length of time after deforestation. The absolute proportion of carbon lost varies considerably, depending on the circumstances, for example, whether land has been cleared for cultivation or pasture purposes. The pattern of loss over time—for example, biomass burning versus trees cut and left to decay—also affects carbon flux. When a tropical forest is converted to pasture or permanent agriculture, the amount of carbon stored in the secondary vegetation is equivalent to the carbon content of the biomass of planted crops, or grass grown on pasture. When a secondary forest is allowed to grow, carbon accumulates, and maximum biomass density is attained after a relatively short time. The greatest loss of carbon occurs during land-use transformation from a primary closed forest to permanent agriculture.

Various estimates have been made of the net amount of carbon released from the soils in conjunction with tropical land-use changes. According to these estimates, biomass carbon losses range from approximately 60 per cent as a result of shifting cultivation to nearly 100 per cent as a result of permanent agriculture. Soil carbon losses after forest clearance are in the range of 20–50 per cent, depending on the land-use conversion (Brown *et al.* 1993; Houghton *et al.* 1987; Trexler and Haugen 1995; Nilsson and Schopfhauser 1995).

Much of the clearing of forests for agriculture in the tropics is cyclic because the practice of shifting cultivation enables reforestation to develop on fallow lands during and after food crop harvests. The period of fallow, after which the forest may be burned and cleared again, may last from 3 to 80 years, according to cultural and environmental conditions. Over the long term, the constant rate of deforestation resulting from shifting cultivation will not contribute to a net flux of carbon into the atmosphere, because the amounts released during burning are offset by carbon accumulation achieved through regrowth (Houghton *et al.* 1987).

In the assessment by FAO (1996), it was estimated that the most common transition of forest cover during 1981-90 was to other land cover, mainly to accommodate cattle ranching and permanent agriculture (49 per cent), followed by short fallow for subsistence agriculture (17 per cent) and fragmented forest consisting of a forest/non-forest mosaic (10 per cent).

These results were based on a quantitative study of pantropical forest cover changes as analysed with a remote sensing survey of the same 118 sample plots from 1980 and 1990 (FAO 1996). The study indicated not only quantitative differences among the three tropical continents, but also strong regional characteristics in the process of change. Tropical forests in Africa are characterized by the transition from closed forests to short fallow. These are defined as temporal agricultural areas with short fallow periods, and thus do not comply with the forest definition. Tropical forests in Latin America are dominated by conversion from closed forests to other land cover. This type of change appears to be the result of centrally-planned operations such as government resettlement schemes, large-scale cattle ranching and hydroelectric projects. Asian tropical forests show two primary conversions: from closed forests to other land use or to short fallow, the former resulting from government resettlement schemes and large plantation programmes (FAO 1996.)

The examination of carbon flows during a land conversion process varies among different studies. For example, Trexler and Haugen (1995), basing their selection on the literature, used the figure of 80 per cent for each continent to represent the biomass carbon released during land clearing. This study is based on the assumption that the characteristic land-use changes within the regions (and subsequent carbon fluxes) given in FAO's Tropical Forest Assessment (1996) will continue over time.

In calculating carbon fluxes, it is important to determine whether fallow forest is permanently taken over as cultivated land or reverted to woodland. In Asia, for instance, this type of change generally follows the sequence—closed forest, long fallow, other land cover and closed forest, short fallow, and other land cover—which reflects the expansion and intensification of shifting cultivation. The assumption here is that most fallow forests are eventually transformed to permanent agriculture or degraded lands. However, as forest areas cleared annually for shifting cultivation have increased (FAO 1996), the areas of fallow forest may not necessarily decrease.

4.3.2 Scenario results

The carbon loss figures for the continents were calculated according to FAO's estimates of the percentage conversion of closed forests to different land cover types[2] and the compiled approximations of carbon losses resulting from these conversions (Brown *et al.* 1993; Houghton *et al.* 1987; FAO 1996). It is estimated that carbon fluxes released from vegetation will amount to 70 per cent of the original quantity in Africa and Asia, and 80 per cent in Latin America, and as a result of deforestation, carbon fluxes from the soil will be 30, 35 and 40 per cent of the original amount, respectively (Table 2.5). The figures are, of course, rough average estimates of actual variations, as no reliable empirical data exist for these variations. However, for this type of aggregate analysis, these are fairly illustrative. Because carbon releases extend over a long timeframe, a constant rate of release was assumed until 2050, totalling 70-80 per cent from vegetation and 30-40 from the soil.

These trends can be regarded as the base scenarios, based on the assumption that the causal mechanism of deforestation prevailing in 1980-90 will continue until 2050. According to the trend scenarios, total tropical forest area could decrease by 590-1,040 million hectares by 2025. This implies carbon fluxes in the range of 80-150 GT (Table 2.6 and Figure 2.4). On the other hand, trend computations produced an interesting result: pantropical deforestation has accelerated slightly from the 1970s to the 1980s (Tables 2.1 and 2.2). In order to be able to calculate trends 3-4, we computed backwards to 1970 from the original year observations with the Chapman-Richard-model, as FAO (1993) had used this model for the 1980 and 1990 updates.

Based on our regression scenarios on deforestation for 1990-2025, some 560-690 million hectares are projected to become deforested and 70-100 GT of carbon to be emitted into the atmosphere (Table 2.6). This analysis indicates that if deforestation could be prevented and the existing tropical forest cover sustained over the same period, the cumulative conserved amount of carbon would match carbon emissions given above. Tropical carbon conservation potential is the greatest in tropical America (40-55 GT C), followed by Africa (21-23 GT C) and Asia (15-17 GT C).

TABLE 2.6
CUMULATIVE DEFORESTATION (MILLION HA) AND CARBON FLUXES (GT C), 1990-2025

	Trexler & Haugen (1		Matsuoka *et al.* (2		This study: Trends		This study: Regression models	
	Deforest-ation	C flux (3	Deforest-ation	C flux (2	Deforest-ation	C flux	Deforest-ation	C flux
Africa	196	12.0-20.6	290	25	181-250	19.0-26.3	198-214	20.8-22.5
Asia	148	11.6-20.4	106	15	130-235	20.2-36.6	97-112	15.1-17.4
Latin America	305	17.4-35.7	304	28	276-558	42.0-84.8	264-362	40.1-55.0
Total tropics	648	41.0-76.7	701	68	587-1043	81-148	559-688	76-95

Notes: (1 1990-2045;
(2 Medium scenario;
(3 Low and high estimates of carbon.

TABLE 2.7
EFFECT OF POPULATION ESTIMATE CHANGE ON FOREST COVER, 2025 (AND 2050)

Region	Latin America		Africa		Asia		Total tropics	
Kind of decrease	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Sum of models 1, 4, 5	Sum of models 2, 3, 6
Population coefficient	-0.71	-1.34	-0.12	-0.31	-0.20	-0.32		
Population change 2015								
millions (1	-17	-17	-26	-26	-92	-92	-134	-134
% (1	-2.9	-2.9	-2.7	-2.7	-3.8	-3.8	-3.4	-3.4
Dependent change 2025 (%)	+2.1	+3.9	+0.3	+0.8	+0.8	+1.2		
Forest area change 2025								
%	+2.1	+2.3	+0.3	+0.5	+0.8	+0.9	+1.4	+1.5
million ha	+11	+15	+1	+2	+2	+2	+15	+18
% of land area	+0.7	+0.9	+0.0	+0.1	+0.2	+0.2	+0.3	+0.4

Note: (1 Change from 1994 edition estimate to 1996 edition estimate in UN World Population Prospects Medium Variant.

Our regression models were calculated on the basis of the 1994 revision of the world population prospects (UN 1995). The 1996 revisions (UN 1998) became available just before the completion of this study and projected a 2.7-3.8 per cent lower population growth by 2015 respectively for the three tropical continents. The sensitivity analysis accordingly reflects the effects of this change on tropical deforestation scenarios. In

Africa, the effects are negligible and in Asia rather modest, but by 2025 (and in 2050) the remaining forest area in Latin America would be 2.1–2.3 per cent higher than reported in Tables 2.4 and 2.7). The deforestation impacts deviate from the percentage declines of population growth prospects due to the variation in the regression coefficients (Table 2.3, row 1).

5. DISCUSSION

5.1 Evaluation of our results

In our opinion, an expansion in natural forest covers during 2025-50 is unlikely, although most tropical forests regenerate naturally (e.g. Kuchli 1997) and successful reforestation may occur without excessive human effort. The scale within which this may be feasible, however, has been exaggerated (e.g. Kuchli 1997). The only pantropical quantitative analysis which we are familiar with, does not indicate any overall natural reforestation, but does suggest a slight increase in Asia with a rather high sampling error (FAO 1996).

Forest plantations in developing countries have doubled from 1980 to 1995, and now total 80-100 million hectares. About half of these are located in the tropics. According to an outlook by FAO (1997), plantation area will increase at least until 2010. However, in our scenarios we are concerned with natural forests only, and it is important to remember that natural and plantation forests are so different in most respects that they cannot compensate each other or maintain similar benefit streams. Plantations have only about half the amount of carbon in their biomass, as compared to natural tropical forests (FAO 1996). Carbon contents of the soil are most likely even lower, because of erosion risk on plantation sites.

We believe that the regression modelling produced sounder scenarios than the trend modelling. This is based on the fact that the regression models have eight independent variables to control the deforestation process, and external scenarios were available for five of these. In the regression modelling, a scientific basis was used to select the future alternatives for the independent variables. If there are deviating views on these prospects, it is possible to rerun the modelling with amended values, whereas the trend scenarios are fixed according to the choice of equation.

The regression scenarios show that deforestation will stagnate after 2025. Economic grounds also support the assumption that deforestation will eventually slow down. As each forest hectare is destroyed, the value of the remaining regions will slowly increase since the agriculturally more productive and accessible areas are converted first, making the conversion of additional hectares less profitable (e.g. Byron and Perez 1996). In the scenarios, economic growth is expected to continue relatively fast (+3.1 per cent per annum; Table 2.4) in Asia, where the critical threshold of US$ 1,500 in Kuznets hypothesis will be surpassed by 2025.

In Latin America, the income threshold point has been already passed. This means that if instead of the zero growth trend (1980-90) assumed here, the ongoing economic growth continues at about 3 per cent per annum which is the 1991-96 average level, this would decrease deforestation pressures (Table 2.3). On the other hand, our assumption of an economic growth in Africa of -1.1 per cent per annum supports the slowing down of deforestation because the income threshold point has not yet been passed in Africa. However, the regression coefficient of GNP is so low (-0.20 in Table 2.3) that the effect of increased deforestation resulting from positive economic growth expectations of 3-4 per cent would remain rather low (Della Senta and Park 1999). In this study, the annual reduction of forest area in Africa is 6.2 million hectares, while a more recent paper on scenarios of African deforestation by Palo and Lehto (1997) with the same logistic function indicated 5.3 million hectares. However, the range of the sensitivity analysis is from 4.9 to 8.1 million hectares. In the 1997 study, the modeling was expanded to 16 independent variables instead of the eight used in this analysis, and also used original year variables instead of updated 1980 and 1990 variables as primary input data. More comprehensive sensitivity analysis on the effects of three alternatives of population and income growth were also conducted.

In another more recent African study (Gaston *et al.* 1998) a more detailed survey and estimation of carbon densities for biomass for both above and below ground were made. They arrived at somewhat lower carbon densities, 82 Mg per ha than the 99 Mg per ha of our study. If this figure by Gaston is the more reliable estimation, it would seem that we have used 17 per cent too high input carbon density.

The Brazilian Amazonian deforestation scenarios calculated by Reis (in this volume) produced a diminishing rate of deforestation already by the year 2010. The result is based on the fact that government policies—for instance, credit subsidies and colonization programmes—have changed. While recognizing the importance of these policy instruments, it was not possible to include these in our modelling. In addition, the recent great fires in Brazil, Mexico, and Indonesia have increased deforestation.

It is interesting to compare the present results to some conclusions from pioneering studies. For instance, Palo and Mery (1986) produced five different trend scenarios for tropical deforestation in 72 countries. The three non-linear scenarios of this 1986 analysis show that by 2025, the remaining forest area will be in the range of 960-1,280 million hectares, an outcome which compares favourably with the present results of 1,070-1,200. The linear scenarios show less dramatic declines in forest areas (1,580-1,620) but Palo and Mery were convinced that the 1986 results were more realistic in relation to the provisional theory of deforestation they launched already in the early stages in the history of deforestation modelling.

It is evident that our carbon flux scenarios, in comparison with estimates by Matsuoka *et al.* (1994), are rather similar for Asia, but somewhat lower for Africa. However, the approximations for Latin America are higher in our study, although deforestation scenarios are in the same fork. The Matsuoka *et al*. study is based the 1980 FAO/UNEP tropical forest resources assessment data (Lanly 1982: 106). However, according to FAO (1995a: 22) the 1980 assessment, as compared to the 1990 assessment, had overestimated total tropical natural forest area by 9.4 per cent. In addition, the scenarios by Matsuoka *et al.* are based on modeling which utilizes only population scenarios while this examination applies eight independent variables, five of which are provided with external scenarios.

The carbon flux estimates by Trexler and Haugen (1995) are low compared with the scenarios conducted here, perhaps because Trexler and Haugen include only vegetation carbon, a fact which underestimates the outcome (Table 2.6). Preferring to use a kind of Delphi-technique, Trexler and Haugen did not apply an objective modelling in their scenario making. They sent about 600 questionnaires to relevant experts in 52 countries in order to assess how a number of relevant deforestation-determining factors would possibly change in the future. A similar questionnaire approach was used by FAO until the 1970s when it was abandoned as unreliable (Persson 1977).

Compared to earlier works by other researchers (Table 2.6), this analysis utilizes the most reliable and up-to-date forest resources data given in the 1990 FAO assessment (1993) and we thus believe that the regression scenarios applied produce very realistic indications of options available to future decision makers. Similarly, our trend forks also map the potential future as base scenarios founded on the assumption that there will be no changes in causal factors. The 1990 assessment also facilitated the use of a large number of observations (578 subnational units in all the tropics) from FORIS-database (FAO 1993).

Our scenario-oriented regression modelling is also quite explicit and the independent variables were selected on the basis of the theoretical considerations in our previous causal modelling of tropical deforestation (Palo and Lehto 1996; Palo and Vanhanen, forthcoming). Our models cope reasonably well with the standard quality criteria (Table 2.3).

5.2 Comparison of deforestation estimates

The most challenging issue regarding the reliability of our results concerns a comparison of values produced by FAO (1997) which re-estimated annual deforestation of tropical natural forests over 1980-90 at 14.6 million hectares and a new respective estimate for 1990-95 at 12.9 million hectares. This would indicate a decelerating pace of deforestation. On the other hand, our data source identified an annual average deforestation rate of 15.4 million hectares for all tropical forests over 1980-90 (FAO 1993). Our regression scenarios produced a respective variable in the range of 16.0-19.7 million hectares from 1990 to 2025 (Table 2.4). Our results indicate higher deforestation than FAO's findings. However, both models indicate a gradual slowing-down of deforestation.

In two subsequent reports, FAO (1997: 10) explains the deviating results for 1980-90 as follows: 'A revised figure of forest cover in 1990 is based on updated population figures for 1990 and some new inventory reports which were available in 1996'. This amendment which could not be considered in our computations may have produced overestimations of a maximum of five per cent compared to the 1990 base year for our scenarios. By reducing our scenario range by five per cent, we arrive at 15.2-18.7 as the annual average for 1990-2025. This is still higher than the FAO approximation of 12.9 for 1990-95 (FAO 1997).

It is possible that the final answer will remain unresolved, but some argumentation can be put forward. FAO computed the 1980, 1990, and 1995 forest area estimates from the original random year national inventory data by means of the so-called Chapman-Richards model (FAO 1993; Scotti, forthcoming). The asymptotic function form will have a stronger decelerating effect the longer the time span in the calculations. The FAO model also uses relative population growth data and it is true that in Latin America and Asia relative population growth rates have been progressively decreasing, while in Africa these have been successively increasing (UN 1998).

In our calculations, we have applied logistic and logarithmic functions (Table 2.3), the form of which deviates from the Chapman-Richards model. The effect of the two function forms appears as 2.6 million hectares as the annual average deforestation scenario from 1990 to 2025 (Table 2.4). We also applied absolute population growth data as the number of inhabitants, which has increased—and will successively continue to do so until 2050 (UN 1998). Both models apply ecological zones, albeit in somewhat different ways: FAO applied lagged forest cover as a third independent variable, which most likely has a prevailing effect, while our examination used GNP and data reliability variables. Consequently in our study, increasing values of these variables are considered to contribute to expanding deforestation until 2025.

The most reliable assessment of pantropical forests and their deforestation were executed as a multi-date satellite image sample survey by FAO (1996). Both this study and the FORIS-database (utilized here), produce almost identical guesstimates on the total 1980 and 1990 tropical forest areas (FAO 1996: 78-9). The simple Pearson correlation coefficient for the two area estimates was 0.969. The respective correlation for the deforestation estimates was clearly only 0.793. From this aspect, it would seem that no strong empirical evidence supports FAO's (1997) calculation of a five per cent reduction in the 1990 forest area, compared to their earlier assessment (1993). It is possible that, if the multi-date satellite survey were to be repeated, the World Forest Resources 2000 Assessment could bring new empirical light to this front, but as of November 1998, this special study had not been fully funded (Palo's interview at the Forestry Department of FAO).

5.3 Future research

The weakest aspect of the present analysis is the transformation of deforestation scenarios into carbon flux scenarios, particularly on the part of Africa. These could be improved by integrating the results of the study by Gaston *et al.* (1998) as inputs into our modelling. Similar studies could also be extended to the other parts of the tropics.

Future research could also include modelling by forest ecosystems. The future of tropical rainforests is the focus of acute international interest, but the present FORIS-database provides information only at the national level, which limits the quality of modelling.

We aim to expand our modelling from the 90 countries included in this analysis to all 115 tropical countries and all 28 nontropical developing countries included in the newly available enlarged version of FORIS-database from FAO. This would include an additional one million hectares of annual average deforestation in our modelling over the period 1981-90. Natural forests and forest plantations should be separately modelled (FAO 1995a, b). Also, we propose to focus future modelling specifically on the ten largest tropical countries with regard to forest area: Brazil, Democratic Republic of the Congo (Zaire), Indonesia, Peru, Colombia, India, Bolivia, Mexico, Venezuela, and Sudan which together cover 65 per cent of all tropical natural forests. Brazil, Indonesia, and the Democratic Republic of the Congo together account for 45 per cent of all tropical forests (FAO 1993). As Brazil and Indonesia have been individually studied (Reis in this volume; Reis and Blanco, forthcoming; Young, as well as Solberg and Purnama in this volume), would be the next relevant country for an individual modelling analysis.

Modelling of deforestation could also be improved from the present examination by expanding the methodology of African deforestation scenarios (Palo and Lehto 1997) to cover the other tropical continents. After a closer residual analysis, instead of the two different models of the current analysis, we would apply only the logistic function form (models 2, 4, 6 in Table 2.3). Also modelling would start from the original random year national forest inventory data, instead of the updated base years' data as in this study. The number of independent variables would also be increased.

Naturally, the modelling of forest transitions and carbon fluxes should be expanded to include all the 179 countries of the world that are covered by FAO's Global Synthesis Assessment (1995a and 1997). This would provide an integrated basis for global forest transition, carbon flux scenarios, and policies. A pilot study has been carried out on this front by Palo and Lehto (1998).

NOTES

* We wish to thank Pekka Kauppi for his advice on carbon flux data sources.

1 However, there are expectations that some stabilization of the world population could happen after 2080 (Lutz 1996: 500).

2 These include plantation, open forest, long fallow, fragmented forest, scrubs, short fallow, other land cover.

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CHAPTER 3

FOREST EXPANSION AND CARBON FLUXES IN THE NORTH

Pekka E. Kauppi

1. INTRODUCTION

Forestry affects the fluxes of carbon dioxide between the atmosphere, vegetation, soils and forest products, and is a potential agent both in contributing to and in controlling the emissions of carbon dioxide. Tans *et al.* (1990), Quay *et al.* (1992) and Keeling *et al.* (1996) have concluded, based on flux measurements of atmospheric gases, that there has been a net flux of carbon dioxide from the atmosphere to land ecosystems in the northern hemisphere. In other words, terrestrial ecosystems in the north have sequestered carbon dioxide from the atmosphere. The land area of China, Europe, Japan, former USSR, and North America totals 5.5 billion hectares. Although forests and other wooded land cover only 2.1 billion hectares of this area (FAO 1995), they can be very significant in the global carbon budget. The carbon pool per unit of land is much larger in forests than in agroecosystems, in alpine and arctic regions, or in deserts.

Dewar (1990) has shown that when forests are managed for the maximum yield of biomass, the contribution to carbon storage from timber products is about 2.5D/T* times the contribution from living trees, where D is the characteristic decay time for reconversion of timber products to carbon dioxide, and T* is the rotation period for maximum sustained yield. When D/T* exceeds a critical value of about 1.0, managed forests and their products together store more carbon than unmanaged forests alone, assuming that soil carbon is not affected. The asymptotic, long-term storage of carbon increases linearly with increasing D.

When forests are converted to agricultural land, the carbon pool is diminished. Also, when old-growth primary forest is cleared and converted into forest which is managed for maximum sustained yield, the standing biomass is reduced by about two-thirds or more (Cooper 1983; Harmon *et al.* 1993). The ratio D/T* in this case is very low, assuming that new forests at the end of the rotation would be similar to -growth forest.

Almost all of the old-growth forests in central Europe were cleared already by the Middle Ages (Kandler 1992). European forests are essentially secondary forests. Also in the United States, settlers had utilized the eastern forests as well as large areas of the western forests by the 19th century. Only a small fraction of land remained as old-growth. After the deforestation phase in the US, forests areas ceased to decline around 1940, and the forests have since then recovered in terms of area, growing stock and the net annual increment (Clawson 1979).

The biomass of secondary forests has increased, because the annual drain (cutting removals plus the boles lost in fellings) has been less than the annual net increment.[1] The analysis of this development must address two critical questions: first, why has the drain (cuttings) been so small?, and second, why has the net annual increment been so large? The following brief remarks are made as a basis for the carbon analysis.

The drain in industrial countries has not increased very much over the last 50 years because oil, natural gas and electricity have replaced the use of wood for space heating, thus reducing demand for firewood. At the same time, forest industries—like all other large industries—have improved their efficiency. More and more products are drawn from one unit of raw material (Ausubel 1996) and the recycling of forest products has been one dimension of this evolution. In conclusion, at least two trends have contributed to a decrease of harvesting: i) replacement of wood by non-renewable materials; and ii) efficiency improvements in forest industry and recycling.

The net increment of North American and European forests has increased in the second half of the twentith century (Clawson 1979; Kuusela 1994). Silvicultural expertise has been widely used for obtaining high timber yields. The objectives of silviculture have in particular focussed on the efficient regeneration of new forest stands and the maintenance of a large growing stock during all phases of stand rotation. All evidence indicates that forestry has been successful in this respect, and the implications are manifold on the fluxes and pools of carbon, both in ecological and economic terms (Hoen and Solberg 1994).

Parallel to the silviculture progress, the prevaling trend in productive agriculture has concentrated on smaller and most productive areas (Waggoner 1994). Less productive land has been abandoned to become available for other uses including forestry. Grazing in particular has become more concentrated, not only because technical efficiency has improved, but because it has been 'difficult to get fockmasters (shepherds) and their families to live and work in isolated conditions in remote areas' (UN-ECE/FAO 1986). Reduced pressures of grazing have promoted forest regeneration and regrowth. In Europe, air pollution has predominately had a positive effect on the growth rate of individual trees, that is, pollution has enhanced forest increment, notably because of nitrogen deposition (Spiecker *et al.* 1996).

Hence, there have been at least four different trends contributing to the increase in net forest increment: i) intensive silviculture; ii) expansion of forest land; iii) reduced grazing pressures; and iv) increased rates of nitrogen deposition.

It follows from this brief analysis that the forest vegetation carbon pool is not in equilibrium but is expanding. The aim of this chapter is to describe and quantify, based on recent reports,[2] this forestry development in western Europe and the US. The general trends are presented and discussed with specific references to the global northern forests, thus revisiting the analysis by Sedjo (1992). An outlook is also presented on the eventual future development of carbon pools.

2. METHODS AND RESULTS

2.1 Vegetation

2.1.1 Europe

Based on the Forest Resource Assessment of 1990 (FAO 1992), Kauppi and Tomppo (1993) have estimated the annual C fluxes related to forest vegetation in seventeen west European countries.[3] Net annual increment and drain were converted to carbon flux estimates by taking into account the density of dry matter in wood, carbon content in dry matter, and the ratio of stemwood to total biomass, including crowns and roots.

The data were based on the official statistics of each country and focused on the year 1990. At the end of the 1980s, only three countries in Europe, notably Austria, Finland and Sweden, monitored net forest increment directly by taking statistically representative measurements of forest growth in the field. The other fourteen countries relied on yield tables and other methods to estimate the net increment. The analysis aimed for the year 1990 but in most cases primary measurements had been conducted during 1985-88. In certain extreme cases, measurements were 10-15 years old, thus referring to forests existing in 1975-80. The estimations of drain were based on industrial and other statistics which were more recent and more comparable between countries.

In all countries the drain was less than the net annual increment indicating an accumulation of vegetation biomass in the forests. The total area of forest and other wooded land in these seventeen countries is 91.2 million hectares. The carbon pool in vegetation is estimated to increase at a rate of 40-60 Tg yr^{-1} (= Teragrams per year = 10^{12} g yr^{-1} = million tons per year). This implies an accumulation of 0.45-0.65 Mg C $ha^{-1}yr^{-1}$, or 45-65 g C $m^{-2}yr^{-1}$. The extremes were Greece and Ireland where the accumulation rates were about + 0 and + 150 g C $m^{-2}yr^{-1}$, respectively. In Ireland, where the forests are mainly young and fast-growing conifer plantations, the drain has been only about one-half of the net annual increment (Kuusela 1994).

2.1.2 The United States

In their analysis Turner *et al.* (1995) refer to conterminous US, excluding Alaska. The data are from forest inventories and are statistically representative (Heath and Birdsey 1993). Methods within the different regions of the US are less heterogeneous than between the seventeen European countries. The landbase in Turner *et al.* (1995) is 200.7 million hectares of timberland, that is, more than twice the size of the area of the west European study. Turner *et al.* (1995) include only forest land, which is defined as land capable of producing 1.4 m^3 $ha^{-1}yr^{-1}$ of industrial wood. The US has a total area of 296 million hectares of forests and other wooded land (FAO 1996).

Living trees in the conterminous United States, according to Turner *et al.* (1995), contain 12,100 Tg carbon. This pool is increasing at rate of 65 Tg yr^{-1}. This implies an average accumulation rate of 32 g C m^{-2} yr^{-1}, which is slightly less than the rate reported for western Europe. The highest accumulation rate within the US is in the northeast region (New England) on private lands.

2.1.3 Russia and Canada

Large areas of forests in Russia and Canada have remained in a state of unmanaged wilderness, a situation different from Europe. In Russia, for example, measures to provide protection against fires and pests cover only 60 per cent of the forests (Nilsson and Shvidenko 1997). Even in the protected zone, large areas are sparsely populated and fires in high forests in these areas are a more common occurrence than in western Europe. Forest fires and other natural disturbances tend to vary greatly in remote areas as a result of diverse climate and forest characteristics (Kurz *et al.* 1995; Kurz and Apps 1996).

The carbon reservoir per hectare of vegetation in boreal forests is approximately the same in the wilderness areas of Canada and Russia than in Scandinavia and Finland, where logging has replaced natural disturbances (Botkin and Simpson 1990; Alexejiev *et al.* 1995; Kauppi *et al.* 1997). Transition from virgin to managed forest may not imply a decrease of the carbon reservoir, although cases of large reductions have been documented (Harmon *et al.* 1990).

2.2 Soil and other forest components

Vegetation carbon as calculated above includes the carbon in roots but excludes the carbon in the soil, forest floor, woody debris, and understory. In the US, the soil pool is estimated to contain 18 200 Tg C, an average of 9.1 kg m^{-2} (Turner *et al.* 1995). In the mature coniferous forests of Finland, the C density in the organic horizon plus mineral soil layer to a depth of one metre is measured to range from 4.0 to 11.9 kg m^{-2}, and there is an additional 1.3-2.4 kg m^{-2} stored in the layer between the one-metre depth and groundwater (Liski and Westman 1995). According to Turner *et al.* (1995), the sum of the additional pools in forest floors, woody debris and understory is 3.2 kg m^{-2}.

Even though the soil carbon pool and other stores are relatively well documented, the eventual rates of change of these pools are poorly known. As there are no direct measurements or statistics on these fluxes, the only way to estimate eventual increases or decreases is to use ecological models and to have an understanding of the ecological processes involved.

Mellillo *et al.* (1993) estimate that the mean net primary production (NPP) of a coniferous forest is 240 and 460 g C m^{-2} yr^{-1} in the boreal and temperate zones, respectively. NPP is the sum of forest increment, mortality, and litter production. In boreal coniferous forests, the production of needle and fine root litter constitutes approximately one-half of the NPP (Albrektson 1980; Persson 1983).

It is reasonable to assume that at least a half of the annual NPP enters the soil in the form of litterfall mainly from roots, needles and branches, adding 120-230 g C m^{-2} yr^{-1} into the soil. An additional accumulation in the order of 70 g C m^{-2} yr^{-1} is released into the soil from removals, as only the stem is utilized for industrial and other uses. Consequently, a total of about 200-300 g C m^{-2} yr^{-1} annually enters forest soil. The interesting question is whether the soil carbon pool is in equilibrium, that is, whether decomposition, leaching, and fire cause a similar flux of 200-300 g C m^{-2} yr^{-1} to be removed from the soil (Olson 1963).

Qualitatively it would appear logical that if the carbon pool in vegetation is increasing and subsequently vegetation sequesters increasing amounts of carbon, then also the carbon pool of the soil must increase and sequester additional carbon. Currently, there is no direct evidence available which could show that this is actually occurring. Let us assume that forest soils in western Europe and the US can accumulate carbon at the same rate as vegetation: 45-65 and 32 g C m^{-2} yr^{-1} in Europe and the US, respectively. The annual input of carbon into the soils is estimated above at 200-300 g C m^{-2} yr^{-1}. An accumulation of 45-65 g C m^{-2} yr^{-1} would imply an output flux from the soil of 140-240 g C m^{-2} yr^{-1}. Ecologically, such a disequilibrium seems reasonable, and an even higher disequilibrium is possible, corresponding to a sequestration rate of, say, 50-100 g C m^{-2} yr^{-1}.

For the landbase of the European study, a rate of 45-65 g C m^{-2} yr^{-1} implies an annual sequestration of 40-60 Tg carbon into the forest soil. Assuming a rate of 32 g C m^{-2} yr^{-1} would yield a slightly greater sequestration in the US, 65 Tg yr^{-1}, because the landbase is larger. However, these estimates are based on ecological reasoning and indirect evidence, without such direct measurements as estimates of forest vegetation.

2.3 Products and landfills

Statistics on wood removal indicate a harvest of about 300 million cubic metres in the seventeen west European countries (FAO 1992). This flux of wood contains about 60 Tg carbon. About one-third is lost during the industrial processes and 40 Tg is transferred annually to the products (Pingoud *et al.* 1996). Removals in the US in 1990 totalled 620 million cubic metres, which on assuming similar technology as in Finland, would imply about 120 Tg being removed from the forests and about 80 Tg being transferred into products annually.

These production figures refer to the input of carbon to the product pool. The output from the pool is the balancing process, that is, degradation and retirement of old product. Changes in carbon stock can be estimated as the difference between input and output. Increases/decreases in the stock can be determined through the collection of data for consecutive years. Based on statistics on Finnish products, Pingoud *et al.* (1996) estimate that the stock in 1990 increased only by 0.6 Tg C yr^{-1}, although the input rate was ten times as high. Landfills are effective accumulators of carbon: the inflow rate is estimated at 3.6 Tg C yr^{-1}, and the net change of the storage at 2.8 Tg C yr^{-1}.

For a sequestration transfer of 0.6 Tg C yr^{-1} to products, the Finnish industry used a flux of 10.5 Tg C yr^{-1} as timber removals. The ratio of product sequestration/removals was 0.6/10.5 = 0.057. Similarly, the ratio of landfill sequestration/removals was 2.8/10.5 = 0.267. Removal estimates are available for western Europe and the US, and an extrapolation is possible, based on the assumption that the Finnish data are a fair representation of western Europe and the US in terms of product mix and usage of products.

Removals in the seventeen west European countries contained annually about 60 Tg carbon (Kauppi and Tomppo 1993). This would imply an accumulation of the product pool of 0.057 x 60 = 3.4 Tg C yr^{-1}; and an accumulation of landfill pool of 0.267 x 60 = 16 Tg C yr^{-1}. In the US with an annual removal of 120 Tg, the respective accumulation would be about 7 and 32 Tg C yr^{-1} in products and landfills, respectively. Additional carbon is imported, especially into the US, as wood products. In the 1980s the fellings in Canada were about 150-170 million cubic metres, of which about two-thirds were exported, mainly into the US. A total of 20 and 50 Tg C yr^{-1} can thus be sequestered in west Europe and the US, respectively, in forest products and as retired forest products in landfills. The results are summarized in Table 3.1

TABLE 3.1
RATES OF CHANGE OF THE MAJOR CARBON POOLS IN WEST EUROPE AND THE UNITED STATES

	Western Europe		United States	
Carbon reservoir	g C m^{-2} yr^{-1}	Tg C yr^{-1}	g C m^{-2} yr^{-1}	Tg C yr^{-1}
Living trees	45-65	40-60	32	65
Soil	45-65	40-60	32	65
Other forest components	-	-	-	-
Products		3.4		10
Landfill		16		50
Total		100-140		190

Note: See text for definitions of the area and other explanations. Estimates for living trees are based on direct measurements; other estimates are indirect evaluations based on expert opinion.

3. DISCUSSION

The forest histories of west Europe and of the US are similar in that land clearance for pasture and agriculture, and the utilization of firewood and other non-industrial wood had earlier resulted in a substantial decline of forest resources. Old-growth forests were mainly harvested during that period. These trends have reversed dramatically, and forest resources, in terms of land area, average growing stock, and net annual increment have increased during the second half of this century at an accelerating rate. The secondary forests are regrowing vigorously, and sequestering a considerable flux of carbon from the atmosphere.

Forest areas in western Europe and in the US total about 0.3 billion hectares, or about 15 per cent of the area constitutes northern forests in the boreal and temperate zone. It can be firmly established from direct observations that the carbon pool of living trees is expanding and sequestering carbon at a rate of about 100-120 Tg yr^{-1}. It is possible that this is an underestimation. Forest inventories are incomplete, and a part of the data dates from the 1970s and the 1980s, and can be lower than for the 1990s.

Based on indirect evidence and ecological reasoning, it is estimated, albeit with uncertainty, that the forest soil of this area can sequester a similar or slightly larger level of atmospheric carbon to the ecosystem than that sequestered by living vegetation. In addition, the carbon pool of forest products and landfills is increasing at a rate of about 50-100 Tg yr^{-1}. Total sequestration, including vegetation, soil, products and landfills in west Europe and the US, can be in the magnitude of 300-350 Tg C yr^{-1}. This is more than Sedjo (1992) estimated for these areas for the years 1985-86.

Critical in this estimate is the uncertainty about the rate of soil carbon sequestration which is believed to be as high or higher than vegetation sequestration. This soil flux, which appears reasonable from an ecological perspective, would imply an annual growth rate of 0.5-1 per cent of the soil carbon pool. This would be about the same as the rate of increase of CO_2 concentrations in the atmosphere. Keeling *et al.* (1995) observe that during the past four decades, there has been a proportionality between rising concentrations of CO_2 in the atmosphere and CO_2 emissions. This implies that as a long-term average, a proportionality also exists between the rate of sequestration and emissions.

If only 15 per cent of the northern forests sequestered as much as 300-350 Tg C annually, the rest would probably sequester less on a per hectare basis. The forest history in Canada and large parts of Russia differs substantially from the one in west Europe and the US; most, if not all, of the trends discussed in this study are lacking. However, fire suppression has been introduced into large areas of these forests, and this may have carbon implications.

On a global scale, it is calculated that afforestation of 460 million hectares could stabilize atmospheric CO_2 concentrations over a period of several decades (Sedjo 1989). Planting new forests, however, is only one of many potential activities within the forest sector (Hoen and Solberg 1994).

Assuming continuity and considering the long-term dynamics of the forest sector, it is easy to predict that the sequestration of carbon will remain effective in the forests of west Europe and the US for at least another 5 to 10 years. However, the sink will probably disappear within 50 to 100 years at the latest because the growing stock cannot expand indefinitely. According to this estimate, forests in west Europe and the US would have compensated for 10-15 per cent of the CO_2 emissions of the area (c.f. Marland *et al.* 1993). Forests have stored carbon with no explicit economic cost attributed to the sequestration.

NOTES

[1] See Kuusela (1994) for western Europe; and Turner *et al.* (1995) for the United States.

[2] In particular, see Kauppi and Tomppo (1993); Turner *et al.* (1995), and Pingoud *et al.* (1996).

[3] The countries include Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Norway, Spain, Sweden, Switzerland, Portugal and the United Kingdom.

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CHAPTER 4

CARBON EMISSION SCENARIOS FOR THE BRAZILIAN AMAZON

Eustáquio J. Reis

1. INTRODUCTION

The accelerated pace of deforestation of the Brazilian Amazon in recent decades is a major source of carbon dioxide emissions into the atmosphere (see Table 4.1). Agriculture in the region represents less than one-fifth of Amazon GDP which, in turn, is less than one-tenth of the Brazilian GDP. In comparing the region's small contribution to the world GDP, these figures suggest that slowing down deforestation of the Amazon can be one of the cheapest alternatives to mitigating the greenhouse effect (Hoeller *et al.* 1991; Nordhaus 1991; Cline 1993).

TABLE 4.8
AMAZON DEFORESTATION AND CO_2 EMISSIONS, 1975-94

	Area		Growth rates p.a.		Carbon missions/year	
Year	10^3km^2	%	10^3km^2	%	10^9t	% world
1975	125	2.6	-	-		
1978	153	3.1	9.3	7.0	0.08	1.1
1988	378	7.7	22.4	9.5	0.20	2.8
1994	470	9.6	15.3	3.7	0.14	1.9

Source: Deforestation in legal Amazon according to INPE; Carbon content of 90.8 t.C/ha estimated by the author.

There are, however, considerable uncertainties with respect to both the future rates of Brazilian Amazon deforestation, and to the costs and benefits of policies to halt it (Pearce 1990; Schneider 1993; Almeida 1992). In an effort to contribute to reducing these uncertainties, this chapter proposes and simulates a model of Brazilian Amazon deforestation and its impact on carbon dioxide emissions to the atmosphere.

To illustrate the potential applications of the model for both projections and policy assessment purposes, the chapter presents simulations for two alternative scenarios for the development of the Amazon region for the period 1985-2010. The scenarios are designed to simulate the effects of the Carajás Investment Programme—a huge mining and steel producing industrial complex located in the southeast of the Amazon region which is expected to have overwhelming economic and environmental consequences—for deforestation and carbon dioxide emissions, in particular (Anderson 1989; Almeida 1986).

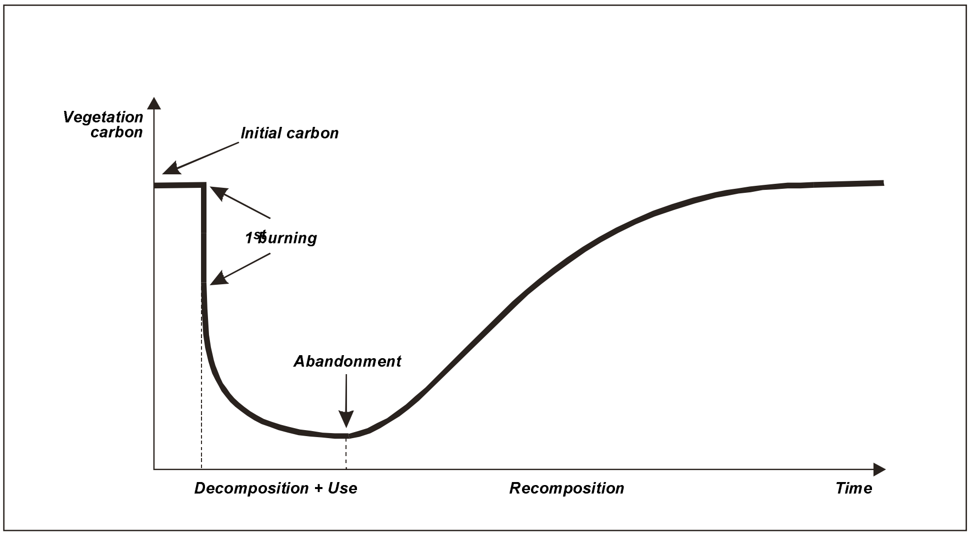
2. THE MODEL

The model consists of two main parts: the first describes the socioeconomic interaction between infrastructure, population, land use, and deforestation; and the second describes the dynamics of carbon stocks in the Amazon vegetation and soils in relation to different land uses.

The socioeconomic interaction was estimated by econometric techniques applied to a panel of data on Amazon municipalities from 1975 to 1985. The main sources of data are the agricultural and industrial census of 1975, 1980, and 1985, and the demographic census of 1970, 1980, and 1990 for all the *municípios* in Legal Amazonia. Methodology and estimation results, however, will not be discussed in this chapter.

Basic assumptions of the socioeconomic model include factors such as i) the sources or proximate causes of Brazilian Amazon deforestation are the uses of land in agropastoral activities; the model assumes that logging is a subsidiary or induced activity; ii) the long run underlying causes of the patterns of land use and deforestation are the expansion of the road network and the patterns of demographic growth; iii) the expansion of road network is an exogenous policy variable which, in fact, can be considered as a proxy variable for government investments in all kinds of infrastructure; iv) the patterns of demographic growth—rural/urban differentials and differences among municipalities—are predetermined by demographic and economic characteristics prevailing in previous periods (Reis and Blanco, forthcoming).

FIGURE 4.5
CARBON CYCLE IN SLASH-AND-BURN AGRICULTURE



The model on the dynamics of carbon stocks in Brazilian Amazon is based on specifications and parameter estimates available in secondary sources. Given the patterns of land use and deforestation, this model quantifies the balance between CO_2 emissions from the burning and decomposition of biomass in above-ground vegetation and soils, as well as the re-absorption of carbon with the growth of secondary vegetation.

The basic assumption of the carbon dynamics model is slash-and-burn agriculture in which three successive phases are distinguished: i) land clearing and forest burning; ii) agropastoral use of land; iii) soil exhaustion, land abandonment, and the recomposition of secondary vegetation in fallow area. Figure 4.1 above illustrates the carbon cycle associated with this pattern of agricultural settlement.

In the following, the whole model is presented in seven sections. The first describes the exogenous growth of the road network, as well as its composition in terms of paved and non-paved roads. The second section specifies the dynamics of urban and rural populations, while the third determines the output of urban activities. The fourth section specifies the determinants of output and the derived demand for land in agropastoral activities. The fifth determines deforestation, as well as its distribution among major types of vegetation as a function of land use in agropastoral activities. The sixth determines the age structure of fallow areas. Finally, the last section models the effects of deforestation and land use on CO_2 emissions from above-ground vegetation and soil.

2.1 Road network expansion

In each municipality, the expansion of federal and state roads are policy variables, exogenously determined by the government. Policy decisions are expressed as the quantity of roads planned in 1985 for each municipality. Planned roads are assumed to be at constant linear rate in time and to be completed within 20 years, that is, up to 2005 (Equation 1 in Appendix).

It should be noted that investments in road network as well as in other kinds of economic and social infrastructure are, to a large extent, endogenously determined by economic activity (Binswanger and Khandker 1992; Pitt *et al.* 1993). To that extent, roads would expand at rates and in directions very different from those originally planned.

Given the extension of roads within a municipality, the model determines the distribution between paved and non-paved roads as function of the geographic density of rural and urban population and output, as well as of the density of other socioeconomic and geo-ecological conditions (invariant with time) prevailing within the municipality, as well as in the neighbouring municipalities (Equation 2 in Appendix).

Thus, road expansion is an exogenous process, but the extension of paved roads is endogenously determined by the 'demand' derived from the socioeconomic conditions prevailing in each municipality. This 'demand' for pavement is satisfied by a partial adjustment process justified by the existence of dynamic costs of adjustment.

2.2 Population dynamics

The rates of growth of rural and urban population are predetermined by the socioeconomic conditions observed in the recent past (that is, the previous census year), both within the municipality and in the neighbouring municipalities. Thus, they are assumed to be a function of: i) population and income per capita in rural and urban areas; ii) transport costs or accessibility conditions, as reflected in the distance to major markets/capitals and in the road network density; iii) socioeconomic infrastructure in urban areas (the per cent of households with access to electricity, water treatment, and sanitation) and land tenure conditions in rural areas (described by the extension of free land available, size of farms, etc.). In addition, they also depend upon the geo-ecological conditions identified by the areas of different types of vegetation and soil, as well as the extension of main rivers crossing the municipality (Equations 3-4 in Appendix).

2.3 Urban output

Urban output is determined by a production function with the following arguments: size of the urban population, socioeconomic infrastructure in urban areas, and accessibility conditions, both within the municipality and in neighbouring municipalities (Equation 5 in Appendix). In addition, the model assumes the existence of economies of scale of urban output in relation to urban population and adjustment costs which give rise to a partial adjustment process (Henderson 1974; Hall 1993; Krugman and Brezis 1993).

2.4 Land use, productivity, and output in agropastoral activities

The model distinguishes the following rural activities or land uses: annual and perennial crops, cattle raising (planted pastures), fallow lands, and logging. Fallow lands are assumed to have no output—open access to natural forests justifies this assumption. Logging is a by-product of agropastoral activities, and therefore is not considered a cause of land clearing. Output and factor prices for all these activities are assumed to be exogenous.

For each agropastoral activity, the derived demand for land and land productivity are estimated as a function of socioeconomic (output and land prices, wages, population and income per capita in urban and rural areas), geo-ecological (like soils, vegetation, geographic area, rivers, etc.), and accessibility conditions (as reflected in road and river network density, ports, distance to major markets) prevailing in each municipality and its neighbours (Binswanger *et al.* 1987). Furthermore, the model assumes adjustment costs which give rise to a partial adjustment process (Equations 6-8 in Appendix).

Land-use decisions are assumed to be hierarchical. Thus, the model estimates and simulates sequentially land-area demand for agropastoral purposes (including both fallow and cultivated areas), for cultivation (crops and pastures), for cropping (annual and perennial), and finally for annual crops.

The output of logging activities is derived from deforestation caused by the expansion of agropastoral activities. In this way, it does not cause deforestation. The volume of logs commercially exploited depends on log prices, the number of lumber industry establishments, road network, urban population (as a proxy for urban activity) and geo-ecological characteristics (like vegetation, soil, distances, etc.—Equation 9 in Appendix). The assumption of no reverse effect of logging on deforestation is particularly justified by the fact that logging data in the IBGE Agricultural Census refer exclusively to the output of agricultural establishments.

2.5 Deforestation

Deforestation is defined as the net change in an area of land, including fallow lands, used for agropastoral purposes. This definition ignores the direct impact on deforestation by other kinds of land use such as logging or other extractive activities, urban settlements, mining, road construction, dams, etc. (Equation 10 in Appendix)

Estimates of the impacts of roads, urban settlements, and dams pose no problem. Major problems, however, arise in the case of logging and placer mining. First, because they are footloose if not illegal activities, thus making it difficult to obtain reliable data on their spatial incidence. Moreover, it is to difficult to asses their direct impact on forest clearing because most of the time they merely degrade forest cover.

Finally, the model allocates deforestation according to two major types of vegetation—forested areas (which include dense and open forest, ecological transition) and non-forested sites (which include savannahs, *campinaranas*, and wetlands)—based on the amount of deforestation, the area available for different types of vegetation and soils, land use in agropastoral activities, accessibility, and socioeconomic conditions (Equation 11 in Appendix).

2.6 The age structure of fallow areas

The absorption of carbon in fallow areas is determined by the biomass recovery (primary productivity) of secondary vegetation that is conditioned by: i) the fito-ecological characteristics of pristine vegetation, in particular rates of carbon accumulation and the climax carbon density; ii) the types of land use the fallow area has been previously exposed to; and, iii) the 'vintage' or average age of fallow, that is, the time elapsed since the start of the fallow period.

To determine the 'vintages' of fallow areas, the model assumes a sequential process for the return of fallow areas to cultivation, according to which older areas are always used before the younger ones. Thus, fallow areas of the vintage (n-1) are only returned to cultivation when fallow areas of vintage n are exhausted. In addition, there is a minimum age required for returning fallow area to cultivation; this is assumed to be the same across the municipalities examined.

Estimation of the age structure of fallow areas in the initial period of simulation is based on two simple arbitrary assumptions: i) the average age of fallow in each municipality is determined by the share of fallow in total agropastoral areas, and ii) by exogenously assuming 2.5 years as the average age of the fallow areas in 1975 for the whole Amazon region.

2.7 Carbon stocks in vegetation and soil

The carbon stock of above-ground vegetation for a given period of time and municipality depends on the average carbon density of the above-ground vegetation which is related to the relative size of the areas of different types of land cover (natural vegetation and agropastoral land uses), as well as to the different 'vintages' of fallow and planted areas (the latter encompass crop and planted pasture areas, but exclude natural pastures).

As shown in Figure 4.1, slash-and-burn clearing causes a sudden drop in the carbon density and is followed by a gradual decomposition of biomass in planted areas and a gradual recomposition of biomass in fallow areas. Thus biomass and carbon densities differ according to the age structure of fallow and planted areas.

Estimation of the carbon budget in the soil for a given period and municipality is made analogously. The only difference in the specification is the absence of the sudden drop in carbon stocks after forest clearing and burning. The present specification of the model does not take into account the increase of carbon stocks in the soil immediately after clearing and burning because this is not relevant in the time unit of the model (five years).

Table 4.2 presents the estimates of biomass and carbon content used in the model for different vegetation or land cover. For above-ground pristine vegetation areas, the model uses estimates compiled by Bohrer (1993) which assume that the dry weight carbon content is 50 per cent of biomass (Fearnside *et al*. 1993; Brown and Lugo 1984). For cultivated areas, the assumption is 5 tons of carbon per hectare (tC/ha) (Bohrer 1993), a low value compared to Houghton *et al.* (1991) who estimate 10 tC/ha for pastures and 5 tC/ha for crops in Latin America.

For forest areas, the model assumes that soil or below-ground[1] biomass is equivalent to 17 per cent of the amount contained in above-ground vegetation (including litter). This assumption is based on estimates for tropical rainforests by Brown and Lugo (1992).[2] This ratio is also used for secondary forests. For savannahs, the model assumes that below-ground vegetation is 1.6 times the biomass contained in above-ground vegetation (Schroeder and Winjum 1994).[3] Finally, for crop areas, including pastures, the model assumes 0.9tC/ha which, according to Sanchez *et al.* 1989, corresponds to a ratio of root/above-ground biomass content of 18 per cent.

TABLE 4.9
BIOMASS AND CARBON CONTENTS FOR DIFFERENT VEGETATION
OR LAND USES (T/HA)

	r [a]	Above-ground plus litter [b]		Soil [c]	
Vegetation/Land use		Biomass	Carbon	Biomass	Carbon
Dense forest	0,5	300	150	51	25.5
Seasonal forest	0,4	186	93	31.6	15.8
Savannahs	0,3	75	37.5	120	60
Ecological Transition	0,4	130	65	22.1	11.1
Wetlands, etc.	0,4	115	57.5	19.6	9.8
Campinarana	0,4	120	60	20.4	10.2
Agropastoral		10	5	16	0.9
Secondary vegetation (climax) [d]		75	37.5	12.8	6.4

Source: [a] Houghton (1992)
[b] Bohrer (1993)
[c] Brown and Lugo (1992); Schroeder and Winjum (1994); Sanchez *et al.* (1989)
[d] Author's estimate.

Note: Observation: r is the rate of carbon decomposition after burning.

2.8 Carbon dynamics in above-ground vegetation and soil

Based on available evidence,[4] the model assumes a 30 per cent combustion efficiency in forest burning. This implies that immediately after burning, 30 per cent of the above-ground carbon is emitted into the atmosphere.[5] In a second phase, the remaining carbon decomposes according to an exponential decay function.

For above-ground vegetation, Houghton *et al.* (1991) estimate that the rates of carbon decomposition after first burning at 0.5 in the case of closed forests, and 0.3 for savannahs (see Table 4.2). These rates imply that 85 per cent and 94 per cent of the above-ground carbon content of forests and savannahs, respectively, will be emitted within five years of forest burning; within 15 years more than 99 per cent of carbon will be emitted in both cases.

Above-ground biomass recomposition in fallow areas is modelled as follows. The biomass content of vegetation at the time of land abandonment is equivalent to that found in crop areas, that is, 5tC/ha crop. In the first phase, due to rapid fast growth of pioneer species, the biomass recomposition of abandoned areas is described by linear rates of growth (Uhl *et al.* 1988; Uhl 1987; Houghton *et al.* 1983). Over time, the diversification of species slows down the recomposition of biomass which converges to a new climax, where the biomass content of the secondary vegetation (forest regrowth) is supposed to be equal to that found in savannah areas, that is, 37.5 tC/ha. It is assumed that fallow vegetation takes 40 years to reach this new climax.[6]

The model assumes that there is no immediate emission of below-ground carbon when forest is burned. Carbon decomposition in the second phase is described by an exponential decay function. The values adopted for the decomposition rates of below-ground carbon are similar to those used for above-ground carbon, that is, 0.5 for dense forests and 0.3 for savannahs.

Finally, the dynamics of below-ground carbon in abandoned areas is analogous to the one assumed for above-ground carbon. Thus, below-ground carbon content starts from the value observed in crop areas (including planted pastures), that is, 0.9tC/ha, and gradually converges to the climax of secondary forests which is assumed to be equal to that of savannah areas, 6.4 tC/ha. This process takes 40 years.[7]

3. SIMULATIONS

3.1 Scenarios

The model is used to simulate the effects of the Carajás Investment Programme on deforestation and carbon dioxide emissions in the period 1985-2010. This huge investment programme includes a three-billion dollar iron ore mining project with a production capacity of 35 million tons per year of sinter feed and pellets; 900 km of railroad connection to sea port; a 4.6 billion dollar hydroelectric dam; and six steel mill projects with a cast-iron production capacity of 800 thousand tons per year.

Carajás is expected to have overwhelming economic and environmental consequences, which include, among others, deforestation, pollution, and poor sanitary conditions in urban areas. The simulations presented, however, refer exclusively to deforestation and the subsequent CO_2 emissions. Moreover, they do not include the deforestation caused by the charcoal requirements of the steel mills.

The direct impact of charcoal production on deforestation, although significant, is presumed to be relatively small and localized when compared to the indirect effects of the expansion of agropastoral activities throughout the whole region induced by the investments in Carajás. Furthermore, a large part of the charcoal supplied to the steel units is residual from logging or agropastoral activities. In 1993, for instance, the share of these activities in the supply of charcoal was 67 and 17 per cent, respectively. In terms of area, an estimate of the deforestation caused directly in 1994 by the Carajás charcoal requirement is 3,857 hectares.

To evaluate the effects of Carajás, simulations of two alternative scenarios—reference and counterfactual—were simulated and compared for the period 1985-2010. The counterfactual scenario assumes that the Carajás Investment Programme does not materialize, implying that the mining, railroad, and steel mill projects will not be implemented, nor will they be developed any time up to the year 2010. In the reference scenario, however, Carajás is effectively implemented with the railroad, mining, and steel mill operations projected to start in 1990. Production is anticipated to develop according to the projected targets of the projects (Rezende and Sampaio 1994).

Simulations of the impact of railroads are based upon the assumption that these are equivalent to the structure of a double extension of paved roads. This is obviously an arbitrary assumption which is only justified by the lack of better information. In addition, the impact of iron ore mining and steel mill projects is simulated as an exogenous increase in the urban GDP of the municipalities of mill locations in equivalent amounts to the increase they bring to the value of output of iron ore or steel production.

Both scenarios assume that all planned roads (at state and federal level) are built within 20 years. Thus, for each municipality, the expansion of the road network from 1985 to 2005 will be similar to the roads planned in 1985. Furthermore, road expansion is made at a constant linear rate and the trend is extrapolated up to the year 2010.

Another important assumption of the counterfactual scenario is that population growth in each municipality is linearly adjusted to make the population growth of entire Legal Amazon equal to the exogenous demographic projections made by IPEA (1991) and Machado (1993). The same parameters are used in the reference scenario but without constraining total population.

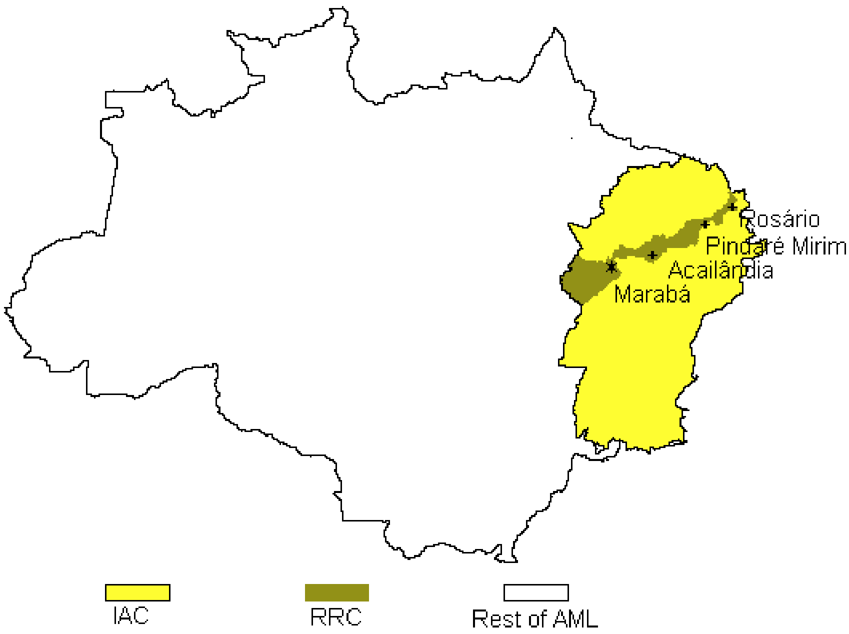
The other exogenous variables of the model are the extension of planned roads in 1985; price of land, cattle, crops, and logs; the value of agricultural credit in relation to output; the cumulative value of SUDAM (the regional development agency) investment credits; the per cent of households with access to electricity, public water and sewerage system; cities with ports exceeding one million tons of turnover/year in 1985

All of these factors are assumed to be equal and constant in both scenarios. This implies that the structure of relative prices across sectors and space is kept unchanged during the simulation period. It also implies that agricultural credit will grow at the same rate as production, and that SUDAM investment credits are completely halted. Finally, the implication for government infrastructure is that it will grow at the same rate as total population.

3.2 Results

Simulations of growth trajectories for the 1985-2010 period for the most important endogenous variables—urban and rural population, urban output, land uses, land yields, and output in major rural activities, deforested areas, fallow areas, and carbon emissions—in the reference and counterfactual scenarios are summarized in Appendix Table A4.1. To illustrate the differential geographical effects of the Carajás Investment Programme, the results of Table A4.1 are presented for three encompassing areas: the 'railroad corridor' (RRC), the 'influence area of Carajás' (IAC), and the whole 'Amazonia Legal' (AML). These are shown in Figure 4.2, together with the location of steel mills.

FIGURE 4.6
MAP OF LEGAL AMAZONIA



Legend: IAC = Influence area
RRC = Railroad corridor
AML = Amazonia Legal

In the reference scenario, deforestation in AML grows 3.2 per cent per annum from 1985 to 2010, compared with the 9.5 per cent per annum observed in the 1980s. The slowdown is largely explained by the urbanization process and the increased turnover of fallow areas. Deforestation, as per cent of geographic area, increases from 8.7 per cent in 1985 to 19.8 per cent in 2010. The scenarios, although worrisome, are not as catastrophic as usually implied by media.

Carbon emissions in the reference scenario decelerate significantly in the simulation period: starting at 345 tons per year in the 1985-90 period, they decline to 167 tons per year in 2005-2010. Estimates of the value of total GDP per unit of carbon (in 1985 dollars) are US$ 52/tC in 1985 and US$ 221/tC in 2010. Equivalent figures for agricultural GDP are, respectively, US$ 7/tC and US$ 29/tC.

Estimated benefits per unit of carbon emissions, therefore, are much higher than the US$ 4/tC usually mentioned in the literature (Nordhaus 1991; Cline 1993). This result is true even when benefits are measured exclusively by the GDP of rural activities. The much higher figures of urban GDP per unit of carbon emitted suggest, however, that the consideration of feedback between agriculture and urban activities can have a decisive effect on the evaluation of benefits.

Finally, the growth of GDP per unit of carbon emitted is impressive. Main reasons include the increasing share of urban activities compounded by their higher productivity levels, the increase of agricultural productivity, and the growing importance of biomass recomposition in secondary vegetation in fallow areas.

Estimates of the effects of Carajás Investment Programme are obtained by comparing the simulation results in the two scenarios. In terms of population aspects, the impact of Carajás is surprisingly small. Indeed, the effects are negligible in the 'influence area of Carajás' (IAC) and Amazonia Legal (AML) while in the 'railroad corridor' (RRC) an increase of 2.5 per cent is projected for the population of the area by 2010. This small demographic effect is explained by the increase of income per capita and the closely-linked the migration to urban areas, as induced by the Carajás Investment Programme. According to the model, both higher income and urbanization tend to decrease rates of population growth.

The effects on GDP are spectacular. Average rates of growth over the 1985-2010 period are 2.3 per cent higher in the 'railroad corridor' (RRC), 1.3 per cent in the 'influence area of Carajás' (IAC), and 0.3 per cent in the entire Amazonia Legal (AML). Therefore, GDP per capita in 2010 will be 70 per cent, 30 per cent, and 7 per cent higher in RRC, IAC, and AML, respectively. But it is important to remember that these are effects on GDP which do not necessarily translate into income to be appropriated by local population. This is particularly true because mining and steel production are capital-intensive activities and most of the profits are remitted outside the region. Naturally, most of the GDP effects accumulate to the urban sector, and the impact on agricultural activities is negligible except in the Carajás corridor (RRC). As a consequence, the deforestation impacts of the programme are relatively small, except in the RRC area where the deforestation area in 2010 will increase 2.5 per cent compared to the counterfactual scenario.

Once again, the surprising results are explained by the strong migration and urbanization process induced by the Carajás Investment Programme. By moving people out of the rural sector and concentrating them around the Carajás Corridor, the programme will promote a reduced demand for land, in particular in the more densely forested areas, and foster the adoption of more labour-intensive techniques.

Finally, in terms of carbon emissions, the impacts of the Carajás Programme are relatively small. For the whole AML, it implies an increase of 1.36 million tons per year, that is, 0.34 per cent increase in the average annual emissions projected in the counterfactual scenario. Compared to GDP this result implies that the Carajás Investment Programme generates US$ 80 worth of output per each additional ton of carbon emitted. Thus, the benefits are extremely high. But as per unit of agricultural GDP, the figure is only US$ 0.15 per ton of carbon emitted.

In concluding, it can be summarized that the simulation shed light on some crucial aspects for projections and policy evaluation of Amazon deforestation. First of all, they point to the need of a systemic approach capable of reconciling a broad geographical perspective with adequate consideration of sector and spatial interactions. Extrapolation of results obtained from specific areas and/or short periods of time to the whole region in the long run should definitely be avoided because effects tend to compensate each other both in time and space. This is particularly true for the analysis of CO_2 emissions where differences both from the diversity of pristine vegetation, as well as from the growth of secondary vegetation in fallow areas can have significant effects.

APPENDIX - MODEL EQUATIONS

(1) $\Delta R_t = R^p 1985 * (t - 1985) / 20$, for $1985 \leq t \leq 2010$

where subscripts j (dropped in most of the cases, for convenience) refers to municipality, t refers to year, and

R_t = road network in year t

$R^p{}_t$ = planned roads in year t.

(2) $SPV_t = f(SPV_{t-1}, Y_{u,t-1}, Y_{r,t-1}, P_{u,t}, P_{r,t,j}, E_{t-1}, W_{t-1}, Z)$

SPV_t = share of paved roads;

$Y_{u,t}$ = value of output of urban activities (includes industry, trade and services) in time t;

$Y_{r,t}$ = value of output of agropastoral activities (including plantation and other forestry activities) in time t, municipality j;

$P_{u,t}$ = urban population in time t;

$P_{r,t}$ = rural population in time t;

E_t = vector of economic variables (like roads, etc.) in time t;

W_t = vector of socioeconomic variables of neighbouring municipalities in time t;

Z = vector of geo-ecological variables (time invariant)

(3) $\hat{P}_{u,t} = f(Y_{u,t-1}, Y_{r,t-1}, P_{u,t-1}, P_{r,t-1}, E_{t-1}, W_{t-1}, Z)$

(4) $\hat{P}_{r,t} = f(Y_{u,t-1}, Y_{r,t-1}, P_{u,t-1}, P_{r,t-1}, E_{t-1}, W_{t-1}, Z)$

where

$\hat{X}_t = \Delta X_t / X_{t-1}$

(5) $Y_{u,t} = f(Y_{u,t-1}, P_{u,t}, E_t, W_t, Z)$

$$(6)\quad A_{k,t} = f(A_{k,t-1}, \Pi_t, P_{u,t}, P_{r,t}, E_{t,j}, W_t, Z)$$

$$(7)\quad \Phi_{k,t} = f(\Phi_{k,t-1}, \Pi_t, P_{u,t}, P_{r,t}, E_{t,j}, W_t, Z)$$

$$(8)\quad Y_{k,t} = A_{k,t} \times \Phi_{k,t}$$

where subscript k (k = 1, 2, ..., f) refers to land uses or agropastoral activities - namely, annual crops, perennial crops, pastures, including fallow lands (denoted by subscript f). Note that logging is excluded from agropastoral activities and also that fallow lands are assumed to have no output.

$A_{k,t}$ = land area use in activity k, year t;

Π_t = vector of prices of outputs, land, and labour year t;

$\Phi_{k,t}$ = land yield in activity k, year t;

$Y_{k,t}$ = value of output in agropastoral activity (excludes logging) in year t;

$$(9)\quad M_t = f(\Delta D_t, P_{u,t}, L_t, \Pi_{m,t}, E_t, W_t)$$

where:

$$(10)\quad \Delta D_t = \sum_k \Delta A_{k,t}$$

M_t = logging volume in year t, municipality j.

ΔD_t = change in deforested area = change in agropastoral land use (including fallow areas)

L_t = output of lumber industry

$\Pi_{m,t}$ = price of logging output

$$(11)\quad Y_{r,t} = \sum_k Y_{k,t} + \Pi_{m,t} * M_t$$

$$(12)\quad \Delta D_{v,t} = f(\Delta D_t, A_{k,t}, P_{u,t}, E_t, W_t, Z)$$

where subscript v (v = 1, 2) refers to forests and non-forests

$$
\begin{aligned}
(13)\quad A_{f,t,i} &= A_{f,t-1,i-1} \text{ if } \Delta A_{f,t} > 0 \text{ and } i < 1 \\
&= \Delta A_{f,t} \text{ if } \Delta A_{f,t} > 0 \text{ and } i = 1 \\
&= \sum_i (A_{f,t-1,i-1} + \Delta A_{f,t}) \text{ if } \Delta A_{f,t} < 0,\ i > i^* \text{ and } \sum_i (A_{f,t-1,i-1} + \Delta F_t) > 0 \\
&= 0 \text{ if } \Delta A_{f,t} < 0,\ i > i^* \text{ and } \sum_i (A_{f,t-1,i-1} + \Delta A_{f,t}) < 0
\end{aligned}
$$

where:

$A_{f,t,i}$ = fallow area of age i in time t in a given municipality

(14) $\Delta A_{f,t} = A_{f,t} - A_{f,t-1}$ = change in fallow between t and t-1

i* = the minimum age required age for the return of fallow area to agropastoral uses, assumed to be the same across municipalities

(15) $A_{f,i,t} * \Delta A_{f,t,i-1} = 0$

(16) $I_0 = K * S_{f,0} / (1 - S_{f,0})$

(17) $S_{f,0} = A_{f,0} / \sum_k A_{k,0}$

where:

I_0 = average of fallow in the initial period

$\Sigma A_{k,0}$ = crop area (including planted pastures and forest plantations) in municipality j in the initial year. K is a parameter determined by the equation

(18) $I_{A0} = \sum_j I_{0,j} * (\sum_i A_{f,t,i,j} / \sum_i \sum_j A_{f,t,i,j})$

where the subscript j refers to municipalities, and:

I_{A0} = is the exogenously assumed age of fallow areas for the whole Amazon region the initial year.

(19) $$C_{a,t} = \sum_v c_{a,v} \times .(A_{v,0} - D_{v,t}) - \sum_v \sum_h c_{a,v} \times A_{k,h,v,t} \times d_{a,h,v} + \sum_k c_{a,k} \times A_{k,t} + \sum_i c_{a,f,i,t} \times A_{f,i,t}$$

where

(20) $$\sum_v D_{v,t} = \sum_v \sum_h A_{k,h,v,t} + \sum_i F_{i,t}$$

where

(21) $$c_{a,f,i} = \frac{M_a}{1 + e^{(a - s_a \cdot i)}}$$

(22) $$d_{a,h,v} = 0.7 \times e^{-r_{a,v} \cdot h}$$

and v stands for vegetation type, h for age or 'vintage' of cropped area, i for 'vintage of fallow areas, k for activity, t for time, and j for municipality and,

$C_{a,\,t}$ = above-ground carbon stock (tons) in time t, municipality j

$c_{a,\,v}$ = above-ground carbon content of vegetation v

$A_{v,0}$ = area of vegetation v in time t in municipality j

$AD_{v,\,t}$ = cleared area in vegetation j, time t, municipality j

c = average above-ground carbon content of municipality j (weighted average of different vegetation)

$AC_{k,\,h,\,t}$= cropped area in activity k, 'vintage' h, time t, municipality j

$d_{a,\,h,\,v}$ = decay factor of above-ground biomass of vegetation j

$r_{a,\,v}$ = exponential decay factor for above-ground biomass in vegetation v

r_a = exponential decay of above-ground biomass in municipality j (area weighted average of $r_{a,\,v}$)

$c_{a,\,k}$ = carbon content of cropped areas (including planted pastures) in activity k, age h

$ck_{\,a}$ = average above-ground carbon content of cropped areas (including planted pastures) weighted by crop areas

$c_{a,\,f,\,i}$ = above-ground carbon content of fallow areas of vintage i

$F_{i,\,t}$ = fallow areas of vintage i, time t, municipality j

M_a = maximum density of above-ground carbon in recomposed vegetation of municipality j

a = $\ln(M_a / ck_{\,a})$

s = $\dfrac{a - \ln(1-e)}{i_{lim}}$

$$(23)\ C_{b,t,j} = \sum_{v} c_{b,v,j} \times .(A_{v,0,j} - D_{v,t,j}) - \sum_{v} \sum_{h} c_{b,v,j} \times A_{k,h,v,t,j} \times d_{b,h,v} + \sum_{k} c_{b,k,j} \times A_{k,t,j} + \sum_{i} c_{b,f,i,t,j} \times A_{F,i,t,j}$$

$$(24)\ c_{b,f,i} = \frac{M_b}{1 + e^{(b - s_b \cdot i)}}$$

$$(25)\ d_{b,h,v} = e^{-r_{b,v} \cdot h}$$

where v stands for vegetation type, h for age or 'vintage' of cropped area, i for 'vintage' of fallow areas, k for activity, t for time, and,

$C_{b,\,t}$ = below-ground carbon stock (tons) in time t

$c_{b,\,v}$ = below-ground carbon content of vegetation v

$A_{v,t}$ = area of vegetation v in time t

$D_{v,\,t}$ = cleared (deforested) area in vegetation v in time t

c_b = average below-ground carbon content per hectare (weighted by areas of vegetation or land use)

$AC_{k,\,h,\,t}$ = cropped area in activity k, 'vintage' h, time t

$d_{b,\,h,\,v}$ = decay factor of below-ground biomass of vegetation v

$r_{b,\,v}$ = exponential decay factor for below-ground biomass in vegetation v

r_b = exponential decay of below-ground biomass (area weighted average of $r_{a,\,v}$)

$c_{b,\,k}$ = carbon content of cropped areas in activity k (including planted pastures, but excluding fallow areas and natural pastures)

$c_{b,\,f,\,i}$ = below-ground carbon content of fallow areas of vintage i

$F_{i,\,t}$ = fallow areas of vintage i, time t

K_b = maximum density of below-ground carbon in recomposed vegetation

a = $\ln(M_b/c_{b,\,k})$

$s_b = \dfrac{b - \ln(1-e)}{i_{lim}}$

APPENDIX TABLE

APPENDIX TABLE A4.1
SIMULATIONS OF THE EFFECTS OF THE CARAJÁS PROGRAMME

	Observations in 1985	Simulations 2010 Basic	Simulations 2010 Potent	Growth rate: 1985-2010 Basic	Growth rate: 1985-2010 Potent	Difference Value	Difference %
			EFC Corridor				
Municipalities	11	-	-	-	-	-	-
Area (km^2)	83,716	-	-	-	-	-	-
Population (inhabitants)							
Rural	704,426	1,771,764	1,546,718	3.76	3.20	-225,046	-12.70
Urban	553,652	1,694,650	2,006,604	4.58	5.29	311,954	18.41
Total	1,258,078	3,466,414	3,553,322	4.14	4.24	86,908	2.51
Roads (km)							
Paved	993	1,475	2,334	1.60	3.48	859	58.24
Non-paved	835	635	1,505	-1.09	2.38	870	137.01
Total	1,828	2,109	3,839	0.57	3.01	1,730	82.03
Land use (ha)							
Temporary crops	205,815	370,234	392,233	2.38	2.61	21,999	5.94
Perennial crops	16,373	49,248	50,737	4.50	4.63	1,489	3.02
Planted pastures	1,129,402	1,852,831	1,962	2.00	-22.45	-1,850,869	-99.89
Fallow land	551,528	1,103,443	1,116,219	2.81	2.86	12,776	1.16
Total agropastoral area	1,903,118	3,375,757	3,460,151	2.32	2.42	84,394	2.50
Unaltered vegetation area	6,468,481	4,995,842	4,911,447	-1.03	-1.10	-84,395	-1.69
Output value (millions US$)							
Temporary crops	34	81	86	3.53	3.78	5	6.17
Perennial crops	3	19	25	7.66	8.85	6	31.58
Planted pastures	41	63	62	1.73	1.67	-1	-1.59
Fallow land	78	164	173	3.02	3.24	9	5.49
Land yield (US$/ha)							
Temporary crops	167	219	220	1.09	1.11	1	0.46
Perennial crops	181	389	490	3.11	4.06	101	25.96
Planted pastures	36	34	33	-0.23	-0.35	-1	-2.94
Fallow land	70	72	74	0.11	0.22	2	2.78
Herd (und)	946,503	2,706,762	3,094,264	4.29	4.85	387,502	14.32
Output value (millions US$)	41	140	155	5.03	5.46	15	10.71
Herd yield (US$/und)	43.00	52.00	50.00	0.76	0.61	-2	-3.85
Graze (Z) (und/ha)	0.83806	1.46088	1.62774	2.25	2.69	0.16686	11.42
GDP (millions US$)							
Agricultural	94	164	173	2.25	2.47	9	5.49
Urban	1,354	3,546	6,242	3.93	6.30	2,696	76.03
Total	1,448	3,710	6,415	3.84	6.13	2,705	72.91
Logging	328,830	1,345,890	1,486,262	5.80	6.22	140,372	10.43
CO_2 emitted (millions ton.)	240	438	449	2.44	2.54	11	2.51
Air (millions ton.)	201	365	375	2.42	2.53	10	2.74
Soil (millions ton.)	39	73	74	2.54	2.60	1	1.37
Municipals with CO_2	11	11	11	0.00	0.00		

(Table continues)

Appendix Table A4.1 (cont.)

	Observations	Simulations 2010		Growth rate: 1985-2010		Difference	
	in 1985	Basic	Potent	Basic	Potent	Value	%
	Influence area of Carajas (IAC)						
Municipalities	156	-	-	-	-	-	-
Area (km^2)	779161	-	-	-	-	-	-
Population (inhabitants)							
Rural	3,384,632	4,414,093	4,084,189	1.07	0.75	-329,904	-7.47
Urban	1,885,312	4,890,078	5,195,916	3.89	4.14	305,838	6.25
Total	5,269,944	9,304,171	9,280,105	2.30	2.29	-24,066	-0.26
Roads (km)							
Paved	4,597	6,007	6,623	1.08	1.47	616	10.25
Non-paved	14,461	19,491	20,606	1.20	1.43	1,115	5.72
Total	19,058	25,498	27,228	1.17	1.44	1,730	6.78
Land use (ha)							
Temporary crops	1,970,616	3,352,396	3,391,059	2.15	2.19	38,663	1.15
Perennial crops	220,741	323,058	321,963	1.54	1.52	-1,095	-0.34
Planted pastures	9,239,513	16,262,797	16,441,793	2.29	2.33	178,996	1.10
Fallow land	9,563,864	15,502,629	15,577,390	1.95	1.97	74,761	0.48
Total agropastoral area	20,994,735	35,440,880	35,732,205	2.12	2.15	291,325	0.82
Unaltered vegetation area	56,921,335	42,475,190	42,183,865	-1.16	-1.19	-291,325	-0.69
Output value (millions US$)							
Temporary crops	365	699	708	2.63	2.69	9	1.29
Perennial crops	73	118	122	1.94	2.08	4	3.39
Planted pastures	335	501	493	1.62	1.56	-8	-1.60
Fallow land	774	1,318	1,323	2.15	2.17	5	0.38
Land yield (US$/ha)							
Temporary crops	185	209	209	0.49	0.49	0	0.00
Perennial crops	333	364	378	0.36	0.51	14	3.85
Planted pastures	36	31	30	-0.60	-0.73	-1	-3.23
Fallow land	80	66	66	-0.77	-0.77	0	0.00
Herd (und)	8,507,834	28,825,162	29,646,235	5.00	5.12	821,073	2.85
Output value (millions US$)	335	1,134	1,166	5.00	5.12	32	2.82
Herd yield (US$/und)	39	39	39	0.00	0.00	0	0.00
Graze (Z) (und/ha)	0.92081	1.77246	1.8031	2.65	2.72	0.03064	1.73
GDP (millions US$)							
Agricultural	909	1,318	1,323	1.50	1.51	5	0.38
Urban	2,317	6,157	8,793	3.99	5.48	2,636	42.81
Total	3,226	7,475	10,116	3.42	4.68	2,641	35.33
Logging	6,020,491	7,477,057	7,652,092	0.87	0.96	175,035	2.34
CO_2 emitted (millions ton.)	1,991	3,857	3,893	2.68	2.72	36	0.93
Air (millions ton.)	1,368	2,468	2,498	2.39	2.44	30	1.22
Soil (millions ton.)	624	1,389	1,395	3.25	3.27	6	0.43
Municipals with CO_2	156	156	156				

(Table continues)

Appendix Table A4.1 (cont.)

	Observations in 1985	Simulations 2010 Basic	Simulations 2010 Potent	Growth rate: 1985-2010 Basic	Growth rate: 1985-2010 Potent	Difference Value	Difference %
			Legal Amazonia				
Municipalities	297	-	-	-	-	-	-
Area (km^2)	4,979,274	-	-	-	-	-	-
Population (inhabitants)							
Rural	6,489,955	8,779,951	8,451,213	1.22	1.06	-328,738	-3.74
Urban	6,399,283	16,378,422	16,688,116	3.83	3.91	309,694	1.89
Total	12,889,238	25,158,373	25,139,329	2.71	2.71	-19,044	-0.08
Roads (km)							
Paved	11,919	19,479	19,953	1.98	2.08	474	2.43
Non-paved	45,935	89,033	90,289	2.68	2.74	1,256	1.41
Total	57,853	108,512	110,242	2.55	2.61	1,730	1.59
Land use (ha)							
Temporary crops	4,933,278	10,237,772	10,273,282	2.96	2.98	35,510	0.35
Perennial crops	923,316	1,857,875	1,855,978	2.84	2.83	-1,897	-0.10
Planted pastures	18,864,820	44,340,513	44,515,229	3.48	3.49	174,716	0.39
Fallow land	19,036,610	39,913,112	39,985,289	3.01	3.01	72,177	0.18
Total agropastoral area	43,758,024	96,349,273	96,629,777	3.21	3.22	280,504	0.29
Unaltered vegetation area	454,169,336	401,578,087	401,297,583	-0.49	-0.49	-280,504	-0.07
Output value (millions US$)							
Temporary crops	1,039	3,025	3,034	4.37	4.38	9	0.30
Perennial crops	348	665	679	2.62	2.71	14	2.11
Planted pastures	691	1,453	1,446	3.02	3.00	-7	-0.48
Fallow land	2,078	5,144	5,149	3.69	3.70	5	0.10
Land yield (US$/ha)							
Temporary crops	211	295	295	1.35	1.35	0	0.00
Perennial crops	377	358	361	-0.21	-0.17	3	0.84
Planted pastures	37	33	32	-0.46	-0.58	-1	-3.03
Fallow land	90	91	91	0.04	0.04	0	0.00
Herd (und)	18,687,129	77,041,932	77,857,135	5.83	5.87	815,203	1.06
Output value (millions US$)	691	3,008	3,041	6.06	6.11	33	1.10
Herd yield (US$/und)	37	39	39	0.21	0.21	0	0.00
Graze (Z) (und/ha)	0.99058	1.73751	1.749	2.27	2.30	0	0.66
GDP (millions US$)							
Agricultural	2,231	5,144	5,149	3.40	3.40	5	0.10
Urban	11,187	33,839	36,478	4.53	4.84	2,639	7.80
Total	13,418	38,983	41,627	4.36	4.63	2,644	6.78
Logging	12,809,749	32,020,968	32,246,490	3.73	3.76	225,522	0.70
CO_2 emitted (millions ton.)	3,970	10,056	10,090	3.79	3.80	34	0.34
Air (millions ton.)	2,664	6,864	6,894	3.86	3.88	30	0.44
Soil (millions ton.)	1,306	3,191	3,197	3.64	3.65	6	0.19
Municipals with CO_2	297	297	297				

Source: Author's simulations

Note: CO_2 values for 1985 are estimated.

NOTES

1 The differences between soil, below-ground, and root biomass in the references are not perfectly clear. As far as possible, data throughout this chapter on below-ground biomass refer to roots, because this is the most dynamic part of below-ground vegetation.

2 Since this value comes from Asian rain forests, Fearnside (1992) suggests instead the factor used by Klinge and Rodrigues (1975). However, Fearnside (1993) uses a similar value (18 per cent). Other references are Manaus, Klinge and Rodrigues (1975),who found that near Manaus below-ground biomass represents 31.4 per cent of total above-ground biomass; Russell (1983: 133) found 23.6 per cent for Jari, PA; and Nepstad (1989) found 10 per cent in Paragominas, PA; the latter is also adopted by Foster Brown *et al.* .

3 This was the only accessible reference on the proportion of biomass in roots and above-ground vegetation for *cerrado* areas; figures are based upon Singh and Joshi (1979) and Fiala *et al*. (1991). For forest areas, similar to Brown and Lugo (1992), they adopted a 20 per cent proportion of biomass in below-ground vegetation.

4 See, for instance, Fearnside *et al.* (1993); Carvalho Jr. *et al.* (1994); Houghton *et al.* (1991).

5 The coefficient of combustion efficiency in tropical forests is usually assumed to be approximately 30 per cent (Houghton *et al.* 1991). However, Fearnside *et al.* (1993) experimental results indicate on average 26.7 per cent of combustion efficiency, while Carvalho Jr *et al*. (1994) finds a value of 25.05 per cent.

6 Odum (1988) and Solomon (1980) suggest the following specification for the recomposition of biomass in abandoned areas: $B_i = \dfrac{M}{1 + e^{(a - s \times i)}}$.

where B_i = above-ground (including litter) biomass content in a give area after i periods of fallow (abandonment) and M = maximum content of above-ground carbon in secondary vegetation of the area.

Assuming B_0 = 5 and M = 37.5, then a = 2,0149 and s = 0,2231.

7 Using the formula in previous endnote (with e = 0.99) these assumptions give a = 1.9617 and s = 0.2217.

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CHAPTER 5

FOREST PLANTATIONS AND CARBON SEQUESTRATION IN CHILE

Gerardo Mery and Markku Kanninen

1. INTRODUCTION

1.1 Climate change and forests

Increases in concentrations of carbon dioxide (CO_2) and other greenhouse gases in the atmosphere have emerged as a major environmental concern because of their impact on global climate and the attendant social and environmental consequences. The alarming increase in the concentration of CO_2—increasing some 25 per cent since the pre-industrial era (IPCC 1992)—has led to a worldwide debate on this issue.

Major reservoirs of carbon are the oceans, the atmosphere, the terrestrial biota, fossil fuels and the soils, and different strategies have been proposed for reducing or controlling atmospheric carbon. Two of these strategies, clearly complementary, are frequently mentioned as efficient ways of tackling the problem: the reduction or limitation of CO_2 emissions and the sequestration of atmospheric carbon dioxide.

The role played by forests in carbon sequestration has been repeatedly highlighted during deliberations on climatic change. Forests are important in the global carbon cycle because they store large quantities of carbon in vegetation and soil, exchange carbon through photosynthesis and respiration with the atmosphere (Brown 1996), and produce products with a high carbon content. However, the carbon stored in forests in the form of wood is released when wood is burned or decays, thus also making forests net emitters of CO_2. Replacing fossil fuels in energy production and using wood in the manufacture of durable products are important approaches to solving the problem (Kanninen 1993). In addition, increasing or at least maintaining permanent forest cover with a high carbon density can contribute to the sequestration of atmospheric carbon into the biosphere. This is especially relevant in view of the fact that above-ground forest ecosystems hold a similar amount of carbon as the carbon content of the earth's atmosphere (Nilsson and Schopfhauser 1995).

Currently, society can adopt different forest management systems that could change the size of terrestrial carbon reservoirs and alter the flow of carbon between vegetation and the atmosphere, and thus have a positive effect on the carbon cycle and climate change.

1.2 Economic role of the Chilean forest sector

Since the beginning of the 1960s, the forest sector in Chile has increased in importance and dynamism, attracting a significant amount of investments. Currently, the sector represents more than 3 per cent of GNP (Cerda *et al.* 1992). The export of Chilean forest products has increased considerably, particularly during the last 15 years, and currently accounts for about 10 per cent of total exports. Total industrial roundwood consumption increased from 4 million m^3 in 1975 to almost 25 million m^3 in 1995 (INFOR 1996). Forest policies implemented in recent decades have mainly supported the establishment of forest plantations for producing industrial roundwood. Fast growing exotic tree species, primarily *Pinus radiata* and *Eucalyptus sp*., have been planted. But the management of natural forests and regulating their rational utilization have been widely neglected.

Several factors confirm the plantation-based forest development approach as an attractive form of land use: i) investments have been highly profitable; ii) risk is confined within acceptable limits; iii) high growth rates for forest plantations of radiata pine and eucalyptus (normally higher than in natural stands) ensure a continuous supply of raw material for the rapidly expanding forest industries; iv) the simplicity of managing and rationalizing the commercial utilization of these artificially created ecosystems introduces important cost reductions; v) good market prospects for products derived from the plantations; vi) forest revenue is realized quickly; and, vii) the government has provided subsidies for forest plantations for more than twenty years. The high rate of carbon sequestration in the plantations could be mentioned as an additional benefit. Accordingly, forest companies can be expected to continue to plant monocultures of fast-growing trees wherever profitable. It should, however, be noted that this approach, which is oriented toward wood production, is mainly motivated by financial and economic issues, disregarding important ecological and social aspects.

The scale of forestry activities and forest industries has rapidly increased during the last thirty years. This was not the result of a unique or specific plan. On the contrary, it was produced by the summation of various, often quite diverse, political decisions. The ultimate result of this development approach is a two-sided forest sector: on one side, we have the dynamic plantation-based forest industries, and on the other side, the backward perception of natural forests and less-than-efficient industry. This dual reality needs to be recognized in the formulation of new forest policies, so that both aspects can be integrated in a balanced manner.

1.3 The purpose of this study

The purpose of this chapter is to analyse the extent of the contribution made by plantations established in Chile during recent decades to the sequestration of atmospheric carbon. While only rough average approximations have been used in earlier papers to determine carbon sequestration, this chapter develops a more detailed approach to estimate the amount of carbon stored in existing radiata pine plantations and the annual flux of carbon sequestration in the Chilean forest plantations. Finally, the role of Chilean forest plantations as a sink of CO_2 is discussed. Consideration is also given to certain social and environmental factors, in particular, the criteria developed internationally to promote sustainable forestry management.

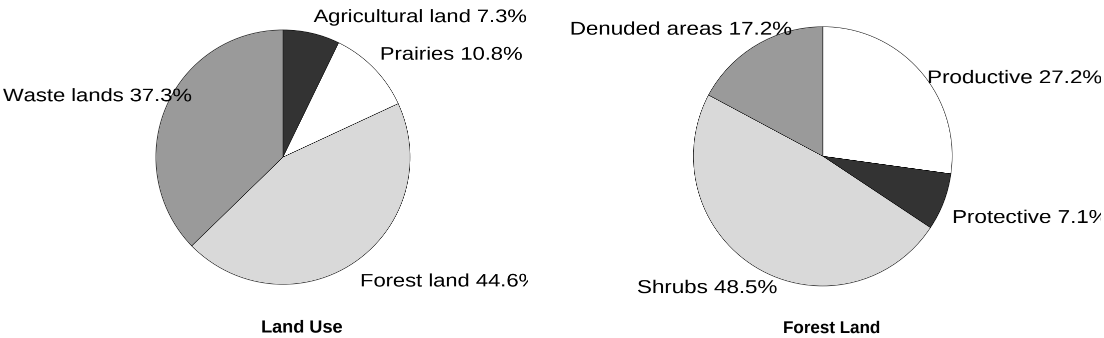
2. FOREST RESOURCES

2.1 Land use

It needs to be mentioned that official figures on forest resources in Chile, particularly those referring to natural forests, should be viewed with some reservation, because a national forest inventory has never been conducted in the country. A national cadastre of the vegetation was initiated in 1994 for the purpose of forest land classification and, to some extent, the evaluation of the state of forests.

Approximately 33.8 million hectares have been classified as forest land, or land suitable for forest growing (Cerda *et al.* 1992). Figure 5.1 indicates the current land-use classification, identifying waste land comprised of mountain peaks, deserts, glaciers, wetlands, and urban areas. Forest land can be classified in four categories (Figure 5.1), one of these being productive forest, which is defined as 'areas dominated by forest tree species which breast high diameter surpass 25 cm and where the solid tree volume exceeds 30 m^3/ha' (INFOR 1990). Productive forests constitute different types of natural forests (81.5 per cent) and forest plantations (18.5 per cent) (INFOR 1996).

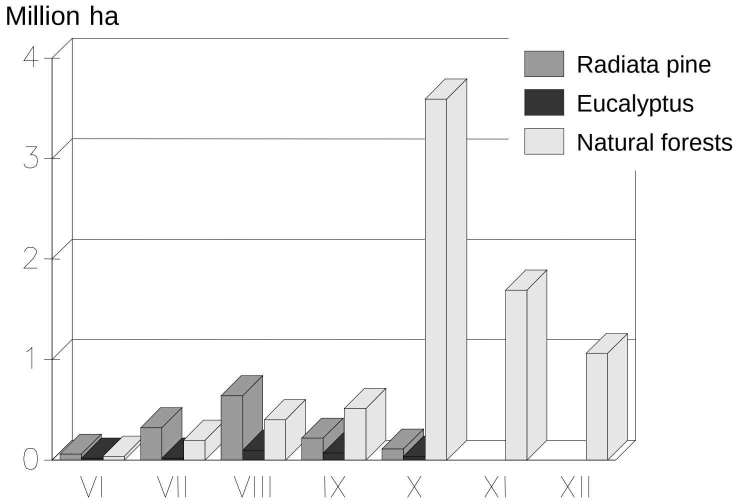
FIGURE 5.1
CURRENT LAND USE AND FOREST LAND CLASSIFICATION



Source: Cerda *et al.* (1992)

The total standing timber volume of potentially productive natural forests in Chile was estimated in 1994 at 891 million m^3 (INFOR 1996). The country also has a rich diversity of natural forests consisting of 12 main forest types and more than 100 tree species (Donoso 1981; Schmidt and Lara 1984). Although several important conifers exist, the most common species are broad-leafed, with the genus *Nothofagus* dominant Donoso 1987; Veblen *et al.* 1981).

FIGURE 5.2
DISTRIBUTION OF POTENTIALLY PRODUCTIVE NATURAL FORESTS AND FOREST PLANTATIONS BY ADMINISTRATIVE REGIONS



Source: INFOR (1996)

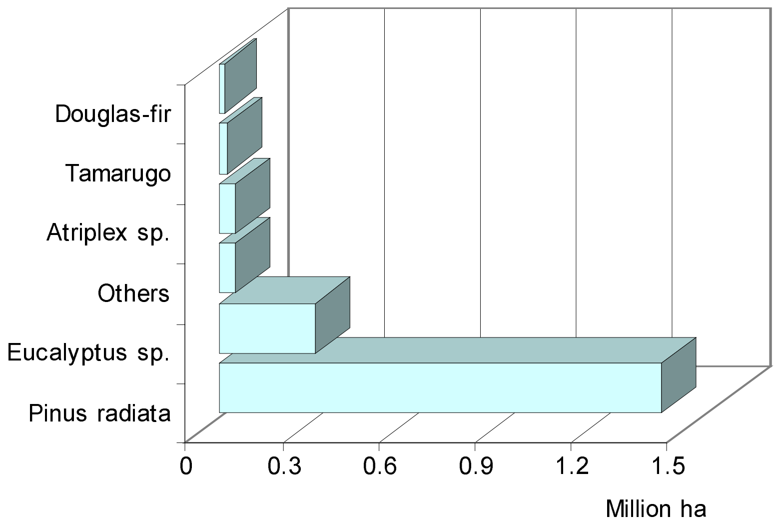
Natural forests vary from arid and semi-arid open sites normally mixed with scrubby vegetation in the northern and central part of the country, to closed rainforests in the south-central and southernmost regions. Over two-thirds of the forests are located in the southern regions (Figure 5.2).

2.2 Forest plantation programmes

Forest plantations began to be established early this century, but large-scale industrial plantations of exotic rapid growing species were not started until the early 1940s. By 1995, about 1.75 million ha of land had been converted to forest plantations with radiata pine (*Pinus radiata*) accounting for 85 per cent of the planted area. Other common species include *Eucalyptus* sp., atriplex (*Atriplex repanda* and *A. numularia*), tamarugo (*Prosopis tamarugo*), Douglas-fir (*Pseudotsuga menziesii*), and poplar (*Populus sp.*) (Figure 5.3). The proportion of non-industrial plantations is quite marginal.

Radiata pine thrives in a broad range of environmental conditions, growing along some 1000 km of Chile's territory, from the sea level up to an altitude of 900 m a.s.l., on soils ranging from sandy to clayish, and with precipitation levels of 400 to 2000 mm yr^{-1}. The adaptability of radiata pine to the climate and terrain conditions prevailing in the central zone of Chile has produced rapid growth that can be roughly estimated to be, on average, 24 m^3 $ha^{-1}yr^{-1}$. Its high yield and numerous uses has resulted in this pine becoming the backbone of the Chilean forest industries and plantations of radiata pine have been established continually over the last twenty years (Figure 5.4). Radiata pine plantations now provide approximately 80 per cent of the total consumption of industrial roundwood. Most pine plantations (more than 98 per cent) are concentrated around the 5th to 10th regions, and region 8 alone contains 47 per cent (Figure 5.2).

FIGURE 5.3
CURRENT AREA COVERED BY FOREST PLANTATIONS (UP TO 1995)



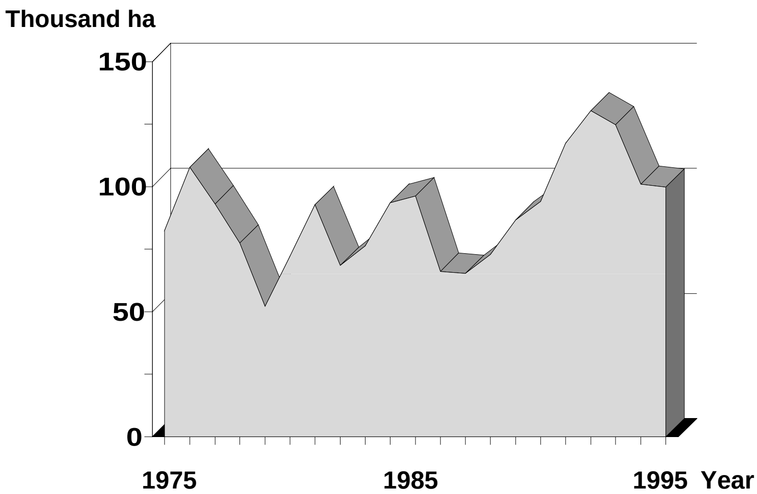
Source: INFOR 1996

Forest plantations have been promoted through subsidies and tax exemptions, and by 1994, close to US$ 150 million had been paid in direct subsidies by the government (INFOR 1996). Legislation introduced in 1974 enabled direct subsidies to be paid up to 75 per cent of the costs for planting, pruning, and miscellaneous managerial expense. As subsidized land was to be re-forested by the recipient, this resulted in significant increases in planted areas over the last twenty years (Figure 5.4), exceeding 65,000 ha annually, mostly in radiata pine. Since 1979, most of the planting operations have been conducted by the private sector, with private companies accounting for over 85 per cent of new plantations (Jélvez *et al.* 1989). Indeed, almost all plantations are owned by private corporations or individuals. The state owns or administers only some 40,000 ha of radiata pine plantations (Cerda *et. al* 1992). More than 40 per cent of the total planted area is owned by three large corporations, each with holdings of approximately 80,000–400,000 ha. Another 40 per cent is in the hands of some 5,000 landowners whose individual holdings range between 50-150 ha, and ownership of the remaining 20 per cent is with medium-sized companies (approximately 30 owners).

Most radiata pine plantations are young, 80 per cent are less than 15 years old (Figure 5.5). As the average rotation age of pine is around 25 years, a considerable proportion of these young stands will mature by 2005-2010. Although different figures on the total volume of harvest becoming available from these pine plantations by the beginning of twenty-first century have been introduced (INFOR 1996 and 1984), an estimate of 23 million m^3 per annum by the year 2005 and 30 million by 2015 seems likely. Even according to a very conservative scenario, future cutting potential for the *Pinus radiata* is enormous. This fact ensures a solid basis for future investments in forest industries.

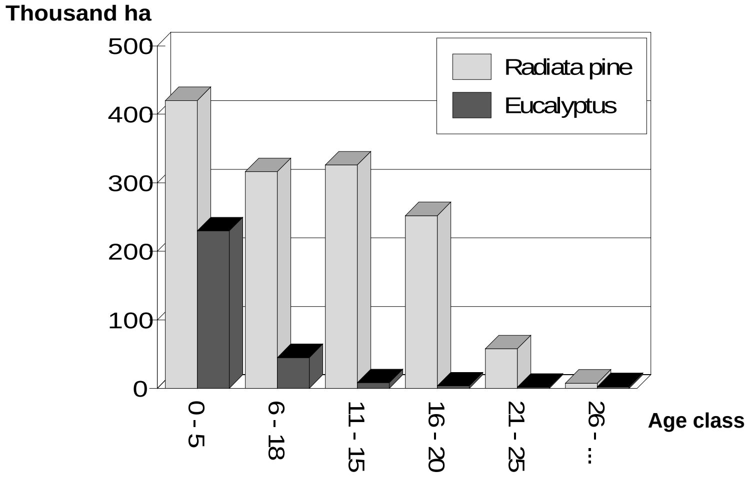
Chile already has the most extensive areas of radiata pine in the world. Moreover, a vast non-productive land area, estimated at 2-5 million ha, has been identified as suitable for forestry operations. It is therefore likely that the total area devoted to forest plantations will gradually increase to 2-3 million ha. In addition, low-yielding farmland currently used for agriculture in the proximity of forest industries is being planted increasingly with forest tree species.

FIGURE 5.4
ANNUAL AREA PLANTED DURING 1975-95



Source: INFOR (1996)

FIGURE 5.5
RADIATA PINE AND EUCALYPTUS PLANTATIONS ACCORDING TO AGE CLASSES, YEAR END 1995



Source: INFOR (1996)

The favourable rate of return on investment in forest plantations is one of the key factors to explain the large investments allocated in Chile for these operations. In a comparative study (1983), Sedjo examined sixteen of the most successful forest plantation cases in different regions of the world. His results showed high profitability for a standard radiata pine plantation in Chile. Also, the special state subsidy programme for promoting forest plantations and increasing knowledge of intensive forestry management have resulted in vast areas being planted annually with radiata pine and eucalyptus.

However, there are actual and potential risks involved with monocultures. These include environmental risks such as severe pests already attacking plantations, and social risks such as rejecting plantations of exotic tree species as a substitution for natural forests. Diversification of the plantations should be encouraged by also introducing other species, either in pure or mixed stands (Lara 1992; Mery 1996).

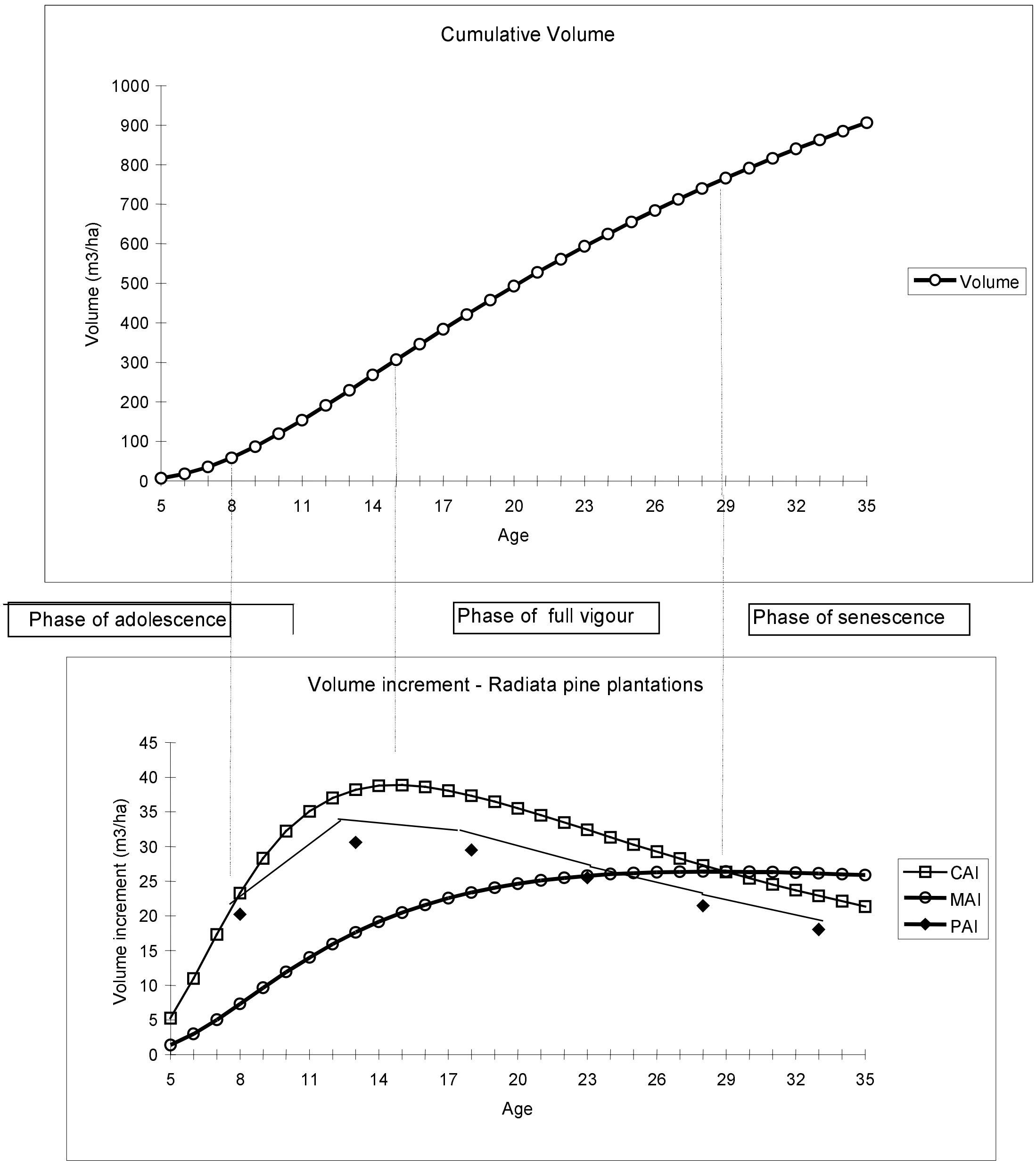
During the last few years, in addition to the popular *Eucalyptus globulus,* different eucalyptus species have been increasingly and successfully introduced to plantations.[1] Auspicious experiences have encouraged private investors to select these new species more often. Also, the exceptional yield of eucalyptus, its demand for many different industrial purposes and its corresponding price on the internal and external markets prophesize rapid expansion for these plantations. Achieving more than 60 m^3 $ha^{-1}yr^{-1}$ in very good sites, the average annual increment of eucalyptus is even higher than radiata pine. Most eucalyptus plantations have been established recently; 95 per cent are less than 15 years old (Figure 5.5). The average rotation age of eucalyptus is often 15-20 years, and a considerable proportion of the current young stands will mature by 2010-15. It is, therefore, expected that by the year 2000, Chile will have close to 400,000 ha in eucalyptus.

2.3 Forest plantations and carbon

The forest plantations of Chile constitute an important source of raw material for the forest industries. Most of the country's man-made forests were established specifically for industrial purposes and most have managed to ensure a steady flow of wood to industrial mills (see section 2.2). Consequently, these plantations are perceived more in terms of crop production than natural ecosystems. They also serve, however, as sinks and reservoirs of greenhouse gases and it should be mentioned that short-rotation forests usually achieve the highest average net annual rates of carbon fixation (Apps and Price 1996).

As of December 1995, Chile had 1.8 million ha of forest plantations, of which 1.4 million in radiata pine, 0.3 million in *Eucalyptus sp.* (INFOR 1996), and 0.1 in other species (Figure 5.3). One of the main aims of the intensive management practice is to achieve full benefit from the forest site by maximizing volume increment. According to common practice, a rotation cycle of 24-28 years for radiata pine and 18 years or less for eucalyptus includes one or two commercial thinnings (at the age of 13-11 and 18-16 years, respectively) and one non-commercial thinning (Hakkila and Mery 1992). Generally, the aim of commonly applied silvicultural management programmes is to produce sawn wood from the last thinning and final cutting, and pulpwood mainly from the commercial thinnings. Obviously a wide variety of management plans adapted to diverse conditions are in operation.

FIGURE 5.6
RELATIONSHIP BETWEEN GROWTH AND VOLUME INCREMENT IN RADIATA PINE PLANTATIONS OF THE IXTH REGION IN CHILE



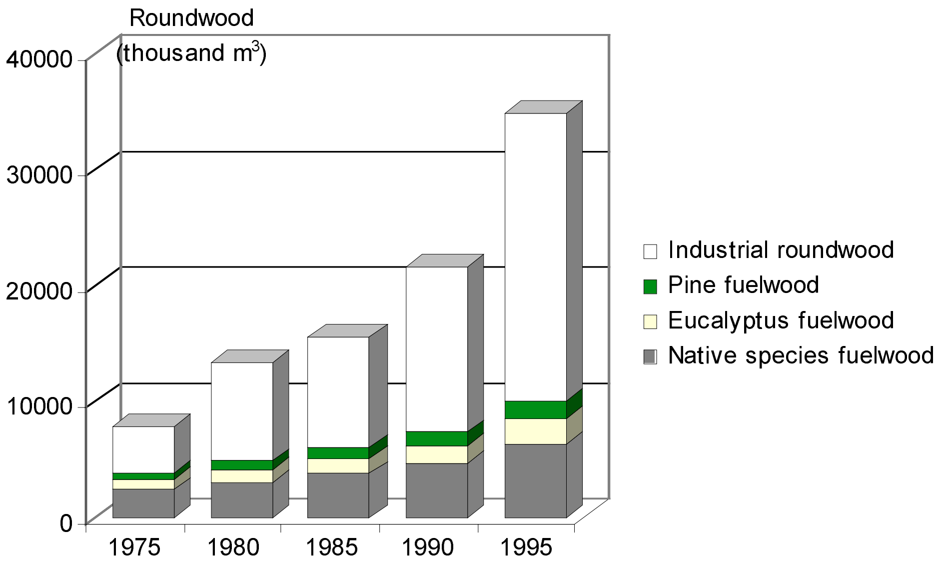
Data source: INFOR 1981

Note: CAI = current annual increment, MAI = mean annual increment, PAI = periodic annual increment.

In view of the fact that the increment curve corresponds to the adolescent and full vigour phases of growth and in view of the typical management programmes described above, it is apparent that silvicultural practices are applied at a time when the physiological activity of trees requires a high amount of carbon sequestration (Figure 5.6). It should be remembered that a forest acts as a carbon sink whenever total harvesting and mortality do not exceed volume increment. Therefore, forest plantations administered according to sound management regimes will constitute a significant and efficient carbon sink. Sustainable logging, by definition, does not have a decreasing impact on carbon reservoirs (Kauppi 1996). Legislation in Chile stipulates that logged-over areas be successfully reforested. Consequently, harvesting and thinning operations result in a temporary reduction of carbon stores—some of which is transferred to forest products—but forests resume carbon accumulation once the stands recuperate from their interrupted development or are regenerated.

Another relevant consideration is the range of products generated from the wood of these fast growing trees. Radiata pine and eucalyptus have been—and still are—mainly used as basic raw material, providing in 1995, for instance, 75 per cent and 14 per cent, respectively of the demand (INFOR 1996). A large proportion of this is consumed by the pulp and paper industry, which as the most important segment of the country's forest industries, constitutes the major bulk of production, investments, and technological input. Annual pulp production in 1995 totalled 2.1 million tons and the paper production 0.6 million tons. On the other hand, the rapidly expanding sawwood industry produced 3.8 million m^3 and production of wood-based panels was 0.6 million in 1995 (INFOR 1996). Other important products are wooden chips and lesser volumes of timber for houses, furniture, agricultural and mining constructions, fruit crates, etc. Moreover, in recent years the range of products manufactured from radiata pine and eucalyptus has been increasing. This is an important development because the production of diverse forest products normally yield a greater potential for carbon storage.

FIGURE 5.7
ROUNDWOOD UTILIZATION IN CHILE, 1975-95



Source: INFOR (1996)

On the other hand, the proportion for fuelwood consumption of the total amount of roundwood (35 million m^3 in 1995) is diminishing in relative terms but is still high (exceeding 10 million m^3 in 1995; Figure 5.7). Forest plantations provide approximately 37 per cent of the total demand for fuelwood (22 per cent eucalyptus and 15 per cent radiata pine) (INFOR 1996). Thus, it can be concluded that radiata pine and eucalyptus plantations are mainly used for producing medium- and long-life products, subsequently preventing the rapid release of sequestered CO_2 into the atmosphere.

It is a fact that carbon sequestration is not the primary aim of forest planting nor the specific goal of the management plans usually applied to forest plantations. It should, nevertheless, be recognized that human beings, through forest management, have the opportunity to increase carbon reservoirs and to modify carbon flows, subsequently contributing to the stabilization of the carbon cycle and to averting climatic change (Brown 1996). Consequently, the carbon issue is becoming a vital element of sustainable forest management.

3. CARBON SEQUESTRATION

3.1 Forests and carbon sequestration

Carbon dioxide is removed from the atmosphere by the photosynthetic activity of terrestrial vegetation and phytoplankton. The two major terrestrial reservoirs are carbon in the soil and carbon in living forest vegetation (Kauppi 1996). Forests, as the most important ecosystem component of vegetation, can be carbon sinks, deposits or sources of CO_2. Forest ecosystem components can be grouped into three major carbon reservoirs: live biomass, detritus, and soils (Apps and Price 1996; Kauppi *et al.* 1996). The dynamics of carbon exchange among—and between—these components and the atmosphere is a basic problem that needs to be understood (Kauppi 1996). In addition, the importance of forest products should also be considered even though they represent only a small fraction of the fixed carbon.

Trees act as a carbon sink when, through photosynthesis, they absorb CO_2 from the atmosphere, subsequently releasing oxygen but retaining carbon. The bulk of the stored forest carbon is in the form of wood (Dabas and Bhatia 1996). Carbon is mainly fixed in the wood as lignin and cellulose. But, the soils and undergrowth vegetation also contain substantial quantities of fixed carbons (Nilsson and Schopfhauser 1995).

The ability of a forest to sequester atmospheric carbon depends on its composition, structure of the stand, genetic characteristics of the trees and plants, and climatic and edaphic factors. Age is another crucial factor. Carbon sequestration and the accumulation of organic carbon in detritus and soil reservoirs change as the stand matures or decays, or whenever natural or man-made disturbances occur. Consequently, the stage of development of a particular forest stand is one of the crucial factors affecting its structure, composition and cumulative biomass volume, and thereby also affecting carbon dynamics. Once a stable distribution in terms of age has been established and if growth conditions remain unchanged, the biomass carbon pools reach a point of steady state when average net biomass C accumulation effectively becomes zero (Apps and Price 1996). The point of the steady state is attained only if the periodic carbon sequestration of forest ecosystems is not exceeded by carbon releases from forest fires and other disturbances over the same period (Brown *et al.* 1996b).

The growth of a tree or stand follows a characteristic S-shaped curve (Figure 5.6). The section between the origin and the first inflection point represents the adolescent age of the plant with a rapid growth rate. The section between the two inflections points represents a faster growth phase of the plant (full vigour phase) when CAI (current annual increment) reaches its maximum. Finally, the section beyond the second inflection point corresponds to the senescence stage when growth gradually declines. At this phase, the mean annual increment (MAI, the ratio of the final growth quantity divided by the number of years) diminishes (Loetsch *et al.* 1973). A stand composed of fast growing young trees will absorb an important amount of CO_2 from the atmosphere, but a younger forest contains less biomass and therefore its carbon pool is smaller. In an old-growth forest, the absorption achieved through growth is largely off-set by wood decay. Left undisturbed, mature forests no longer sequester atmospheric carbon but are able to retain a fixed amount of carbon within the vegetal tissues, acting as carbon deposits (Kanninen 1993; Apps and Price 1996).

Forests and other types of vegetation covers become sources of CO_2 when disturbed and the carbon appropriated earlier is released into the atmosphere. Natural or anthropogenic in origin, disturbances include deforestation or the conversion of forests to non-forest uses such as agriculture land and pastures; unsustainable logging or degradation that occurs from damage to residual trees and to the soil through wild fires, poor logging practices, overgrazing, excessive fuelwood gathering and other events; diseases and decay. Even though logging reduces the amount of carbon stored per unit area, logged forests can be regenerated under sound management regimes to accumulate carbon again (Brown 1996).

At the global level, it is estimated that forests are the basic net source of carbon dioxide and other greenhouse gases like methane (CH_4), carbon monoxide (CO) and nitrogen oxides (NO_n) to the atmosphere (IPCC 1995). The forest also constitutes a vital link in the plant-to-soil sink mechanism, acting as a pump which discharges some of the carbon contained in the litter into deeper soil layers where it is finally stored (Kauppi 1996). The carbon pump phenomenon varies widely according to the inherent characteristics of the different ecosystems. But the vast areas being deforested and degraded every year (FAO 1997) cannot be compensated adequately with new forest plantations and/or increased quantities of biomass per unit area as proposed by current management programmes.

3.2 Estimation of carbon pools, fluxes and balance

3.2.1 Estimation procedure

To estimate the carbon sequestration of forest plantations in Chile, two basic assumptions have to be taken into consideration. An important amount of carbon is already stored in the existing radiata pine and eucalyptus plantations that had been established prior to December 1995. The increment of this stored carbon pool is a cumulative phenomenon. See section 3.2.3 for an estimation of this carbon pool.

The annual flow of carbon is calculated on the basis of both existing and new plantations. The plantation volumes reduced by harvesting, forest fire, or disease are subtracted from the respective volume of growing stock. However, thinnings have not been taken into consideration nor the amount of carbon stored in the soil and forest products. See section 3.2.2 for this estimation.

3.2.2 Estimation of the stand volume and volume increment

The total volume of the growing stock was estimated on the basis of the total area of forest plantations as of December 1995 (Table 5.1) and the mean cumulative volume which for radiata pine plantations was estimated with the yield functions (see equation 1 and Table 5.2) developed by Instituto Forestal (INFOR 1981) for each administrative region. These functions estimate the average values of a stand's stemwood volume, and have been calculated on the basis of numerous sample plots. The dependent variable of these models is the cumulative stand volume (V, $m^3\ ha^{-1}$) and the independent variable is the stand age (E, years). (For the values of parameters ***a*** and ***b*** see the Table 5.2):

(1) $\quad \text{Ln } V = a - b\,(1/E)$, where V = stand volume ($m^3\ ha^{-1}$), and E = stand age (years).

TABLE 5.1
TOTAL PLANTED AREA ESTABLISHED BY THE END OF 1995
BY AGE CLASSES AND ADMINISTRATIVE REGIONS ('000 HA)

	Age classes (in years)						
Region	0 -5	6-10	11-15	16-20	21-25	26-...	Total
Radiata pine							
V and RM	4	3	3	3	2	1	17
VI	21	10	11	15	4	0	60
VII	78	90	92	36	24	1	320
VIII	192	135	147	147	19	3	640
IX and X	125	78	73	52	9	2	337
Sub-total	420	316	326	253	58	7	1374
Eucalyptus							
All regions	229	44	7	3	3		286
Total	649	360	333	256	61	7	1660

Source: INFOR (1996)

TABLE 5.2
PARAMETER VALUES OF EQUATION (1) Ln V = a - b (1/E)
BY ADMINISTRATIVE REGIONS

Region	Parameter values *a*	Parameter values *b*
V and RM	6.909100	26.7026
VI	7.294867	29.3753
VII	7.608300	32.1089
VIII	7.282900	23.8646
IX and X	7.621200	28.4121

TABLE 5.3
MEAN CUMULATIVE STEM VOLUME (M^3 HA^{-1}) IN FOREST PLANTATIONS
ACCORDING TO AGE CLASSES AND ADMINISTRATIVE REGIONS

Region	Mean cumulative stem volume (m^3 ha^{-1}) Age classes (in years)					
	0-5	6-10	11-15	16-20	21-25	26-...
Radiata pine						
V and RM	5.0	35.6	128.4	227.2	313.6	411.2
VI	5.0	37.5	153.7	288.0	410.6	553.2
VII	5.0	36.4	170.4	338.5	498.8	690.9
VIII	6.0	73.7	232.1	386.5	515.6	656.8
IX and X	5.0	58.5	229.4	421.1	593.4	791.7
Eucalyptus						
All regions	2.0	35.3	135.4	285.7	517.0	N.A.

TABLE 5.4
TOTAL STANDING STEMWOOD VOLUME OF THE GROWING STOCK BY YEAR END, 1995
ACCORDING TO AGE CLASSES AND ADMINISTRATIVE REGIONS (MILLION M^3)

Region	Age classes (in years) 0-5	6-10	11-15	16-20	21-25	26-...	Total
Radiata pine							
V and RM	0.02	0.11	0.41	0.66	0.66	0.48	2.34
VI	0.10	0.38	1.66	4.23	1.51	0.22	8.10
VII	0.39	3.27	15.61	12.11	12.07	0.79	44.23
VIII	1.15	9.95	34.21	56.64	9.90	1.72	113.56
IX and X	0.62	4.59	16.72	21.90	5.27	1.59	50.70
Sub-total	2.28	18.3	68.61	95.54	29.41	4.8	218.93
Eucalyptus							
All regions	0.46	1.56	0.99	0.78	1.41	0.0	5.19
Total	2.74	19.86	69.60	96.32	30.82	4.79	224.12

The functions were applied for calculating the cumulative volume of radiata pine (Table

5.3). The total stemwood volume of radiata pine plantations was 219 million m^3 by the end of 1995 (Table 5.4).

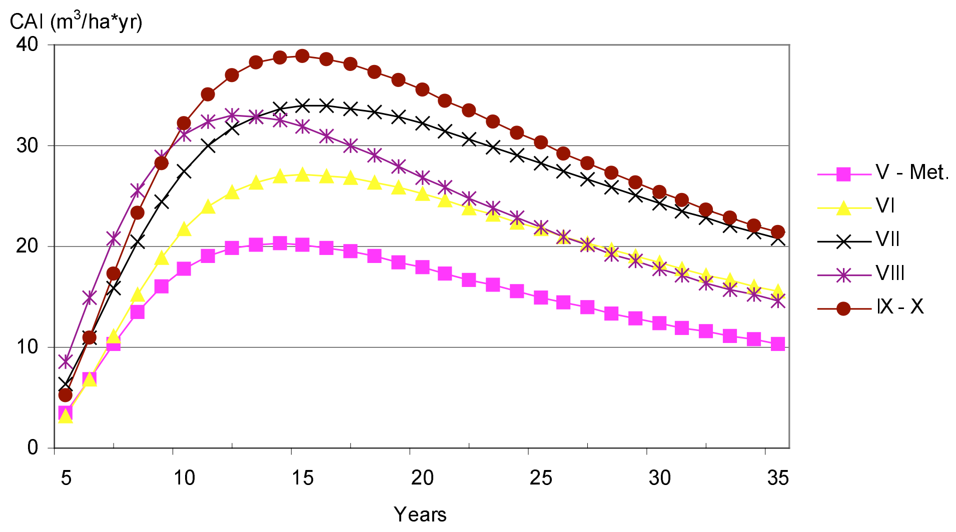
Unfortunately, it was not possible to apply the same methodology to eucalyptus plantations, because yield functions by administrative regions were not available. Instead, the mean cumulative volume (Table 5.3) was estimated, using a growth model developed for the *genus eucalyptus* in Chile (INFOR 1994). We assumed that in an 'average site' and with an 'average management regime', an eucalyptus stand reaches an average height of 28 metres in 20 years and that the stand has a density of 1,600 seedlings ha^{-1}. The total standing volume of eucalyptus plantations in 1995 was 5.2 million m^3 (Table 5.4).

Volume increment: The yield functions explained above (Equation 1) were applied for calculating the current annual volume increment (CAI) (m^3 ha^{-1} yr^{-1}) for radiata pine

TABLE 5.5
CURRENT ANNUAL VOLUME INCREMENT (M^3 $HA^{-1}YR^{-1}$) BY THE MEAN VALUE OF THE AGE CLASSES AND ADMINISTRATIVE REGIONS, 1996

	Age classes (in years)					
Region	5	8	13	18	23	28
Radiata pine						
V and RM	3.5	13.5	20.2	19.0	16.1	13.4
VI	3.2	15.3	26.4	26.4	23.2	19.7
VII	6.3	20.5	32.9	33.3	29.9	25.8
VIII	8.6	25.6	32.9	29.0	23.8	19.3
IX and X	5.3	23.3	38.2	37.3	32.4	27.3
Eucalyptus						
All regions	2.0	11.7	25.3	32.0	33.4	n.a.

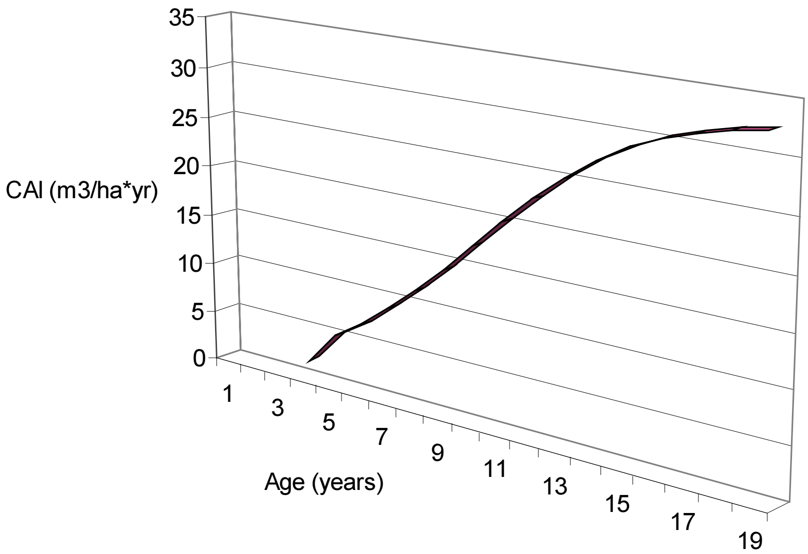
FIGURE 5.8
CURRENT ANNUAL VOLUME INCREMENT (CAI) (M^3 HA^{-1} YR^{-1}) OF *PINUS RADIATA* PLANTATIONS IN CHILE BY ADMINISTRATIVE REGIONS



and the mean annual volume increment (MAI) (m^3 ha^{-1} yr^{-1}) by age groups and by administrative regions. As was explained above, estimation by administrative regions and age is relevant because important differences in the figures for volume growth and volume increment exist between regions, as Table 5.5 and Figure 5.8 indicate.

Due to the lack of data, the above method was not applied to eucalyptus plantations. Instead, the current volume increment given in Table 5.5 and Figure 5.9 was calculated with the 'average site' and 'average management regime' method explained above. It is possible that this may be an under-estimation of growing stock for plantations exceeding 15 years, because the growth functions used were defined for plantation maturity less than 15 years only.

FIGURE 5.9
CURRENT ANNUAL VOLUME INCREMENT (CAI)(M^3 $HA^{-1}YR^{-1}$)
OF *EUCALYPTUS SP*. PLANTATIONS IN CHILE



3.2.3 Estimation of the current carbon pool and annual flux

Total carbon pool: To estimate the total amount of carbon stored in the stemwood of trees (Cpool) the total standing volume is multiplied by the average wood density and by the estimated carbon content of the biomass (Hollinger *et al.* 1993). However, carbon is stored not only in the stemwood, but also in branches, foliage, roots, forest floor litter and in the vegetation growing in the understorey. Various studies have been carried out for estimating the conversion of stemwood biomass to total biomass. Here, we use the models developed by the New Zealand Forest Research Institute for their local *Pinus radiata* stands. The model is based on studies on forest biomass and it takes into consideration the effect of tree age, plantation density and site fertility (Ministry for the Environment 1994; MacLaren *et al.* 1994) (Equation 2). Thus, the carbon pool fixed in the total biomass of radiata pine and eucalyptus plantations, Cpool, is given by

(2) $$Cpool = V \bullet \rho \bullet c \bullet \alpha$$

where:

V = total standing volume as estimated in Table 5.4 (m^3)

ρ = 4.29 10^5 g m^{-3} for radiata pine, average wood density (dry matter) (Perez 1983)

ρ = 6.23 10^5 g m^{-3} for *Eucalyptus globulus*, average wood density (dry matter) (Perez 1983)

c = 50 per cent carbon content of the biomass (Hollinger *et al.* 1993)

α = 1.89 the ratio of total biomass to stem biomass (Hollinger *et al.* 1993).

<u>Annual fluxes of carbon</u> are usually calculated on the basis of the increment of the carbon pool and its decrease due to removals and mortality. In our case, available data included the net area of forest plantations. A rough estimate of the annual flow of carbon to forest plantations in 1996 is obtained by summing the carbon flux to existing plantations (S1, Equation 3) and the flux to newly established plantations (S2, Equation 4). Thus, S = S1 + S2. In 1996, the total planted area increased by 64,000 ha, a figure which also includes reductions due to final cuttings and forest fires. S1 and S2 were calculated for both species as:

(3) $S_1 = (CAI \bullet \rho \bullet c \bullet \alpha \bullet a_o)$

(4) $S_2 = (CAI \bullet \rho \bullet c \bullet \alpha \bullet a_n)$

where:

S_1 = annual carbon sequestration in the area planted up to the end of 1995

S_2 = annual carbon sequestration in the new planted area in 1996

CAI = current annual increment in m^3 ha^{-1} yr^{-1} for radiata pine and eucalyptus (Table 5.5)

ρ = 4.29 10^5 g m^{-3} for radiata pine, average wood density (dry matter) (Perez 1983)

ρ = 6.23 10^5 g m^{-3} for *Eucalyptus globulus,* average wood density (dry matter) (Perez 1983)

c = 50 per cent carbon content of the biomass (Hollinger *et al.* 1993)

α = 1.89 the ratio of total biomass to stem biomass (Hollinger *et al.* 1993)

a_o = planted area up to the end of 1995 (ha)

$a_n =$ planted are during the year 1996 (ha).

The values of current annual increment given in Table 5.5 have been applied for estimating the annual carbon sequestration (S) during 1996. It should be stressed that *CAI*, ρ, *c* and α may vary with species, tree age, forest location and edaphic factors, and silvicultural regimes.

3.3 Carbon pools, fluxes, and balance

Using the Equation 2, we obtain a total carbon pool of approximately 88.8 Tg (tera-gram or million tons) in radiata pine plantations or 64.6 ton ha^{-1}. The carbon pool estimated in *Eucalyptus sp* plantations is 3 Tg. Therefore, forest plantations established in Chile up to December 1995 constitute a carbon pool close to 92 Tg (Table 5.6). This is equivalent to 48.8 tons ha^{-1} of carbon stored in forest plantations.

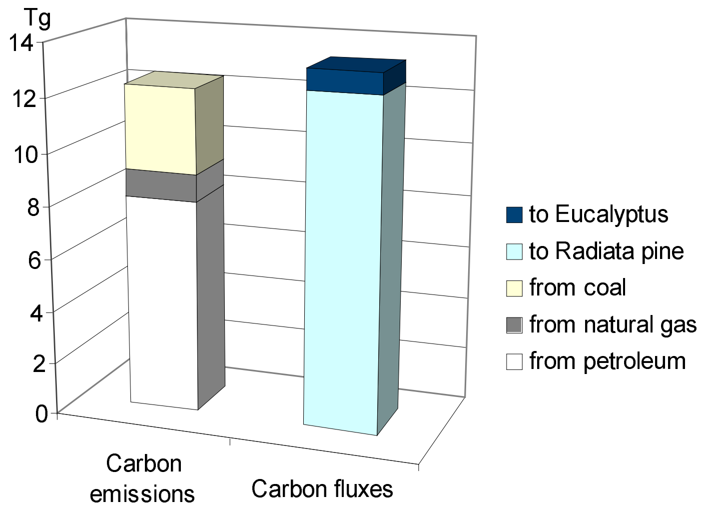
TABLE 5.6
CARBON POOLS AND FLUXES IN CHILEAN FOREST PLANTATIONS

	Age classes (in years)						
	0-5	6-10	11-15	16-20	21-25	26-...	Total
Cpool (Tg)							
Radiata pine	0.93	7.42	27.82	38.73	11.92	1.94	88.75
Eucalyptus	0.27	0.92	0.58	0.46	0.83		3.06
Total	1.20	8.33	28.40	39.19	12.75	1.94	91.81
Flux S1(Tg yr-1)							
Radiata pine	1.17	2.97	4.46	3.17	0.64	0.06	12.47
Eucalyptus	0.27	0.30	0.11	0.05	0.05		0.79
Total	1.44	3.27	4.57	3.22	0.70	0.06	13.26

Applying Equations 3 and 4, we have S1 = 12.5 Tg yr^{-1}. S2 is negligible because the value of CAI is close to 0 during the first year. So, S2 can be excluded from the table of results. Thus, we estimate that the total carbon flux is 12.5 Tg yr^{-1} for the radiata pine plantations in 1996. This is equivalent of 9.1 ton ha^{-1} yr^{-1}. In the case of eucalyptus plantations the total carbon flux (S1) for the year 1996 is 0.8 Tg yr^{-1} (Table 5.6), which is equivalent to 2.8 ton ha^{-1} yr^{-1}. Consequently, the total flux of atmospheric carbon for forest plantations in Chile was approximately 13.3 Tg yr^{-1} in 1996. The values obtained here for the annual carbon sequestration are higher than the values reported by other researchers (Mellillo *et al.* 1993, Dixon *et al.* 1994). Considering that the carbon content in the molecule of CO_2 is 27.3 per cent, our calculations indicate that 48.7 Tg yr^{-1} of atmospheric CO_2 is sequestered annually by the Chilean forest plantations.

In comparing the estimates presented above with the carbon dioxide emissions in Chile for 1996, one can conclude that annual carbon fluxes to forest plantations exceeded annual carbon emissions which were estimated as 12.33 Tg (EIA) (Figure 5.10) in that year. The emissions are estimated from fossil fuel consumption of petroleum, coal and natural gas. Therefore, forest plantations have a positive role in the carbon balance of the country because they contribute to absorbing excessive CO_2 emissions moving into the atmosphere, and thus preserve the equilibrium between different components of the carbon cycle. Carbon balance is crucial for maintaining the ability of the earth's biological ecosystems to provide the goods and services that are essential for sustainable economic development.[2]

FIGURE 5.10
BALANCE BETWEEN CARBON EMISSIONS FROM FOSSIL FUEL CONSUMPTION AND CARBON FLUXES TO FOREST PLANTATIONS IN CHILE, 1996



4. THE CHALLENGES OF SUSTAINABLE FORESTRY AND CARBON SEQUESTRATION

4.1 Scenarios for carbon sequestration in radiata pine plantations

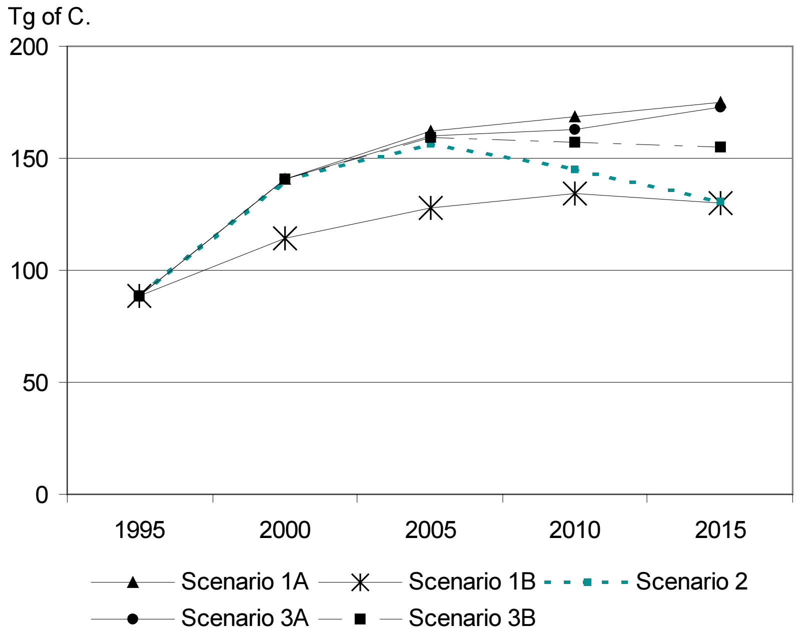
Forests have the potential to contribute to climate change through their influence on the global carbon cycle. In particular, the importance of local measures to promote carbon sequestration and conservation has been emphasized (Brown 1996). Here, we present some scenarios on the potential development of carbon pools in the radiata pine plantations up to the year 2015. Unfortunately, the lack of forestry statistics, especially for cuttings, makes the development of these scenarios difficult. Eucalyptus plantations could not be included in these scenarios, due to lack of reliable data and uncertainties regarding the future development of these plantations.

The five scenarios developed in this study belong to three 'groups': Group 1 (scenarios 1A and 1B) in which future plantations (both reforestation and afforestation) are established at the rate attained in recent years, i.e. approximately 63,500 ha yr^{-1}, resulting in a 10-20 per cent increment of total plantation area by the year 2015. Group 2 (scenario 2) assumes that the current plantation area is maintained, and only the harvested areas are reforested; no afforestation is carried out. Group 3 scenarios (3A and 3B) is based on the assumption that future plantations (both reforestation and afforestation) are established at approximately 31,750 ha yr^{-1}, which is half of the initial rate of recent years, and later at a rate to compensate areas harvested. This would mean an increment of 35-46 per cent of the total plantation area by 2015, based on the assumption that the government is willing to continue its support for creating new areas through subsidies for afforestation, but not for reforestation.

For two groups (1 and 3), we developed different sub-scenarios (A and B) to illustrate the impact of different forest management options on carbon pools. In subclass 1A, all plantations older than 26 years and 50 per cent of plantations in the age group 21-25 years are harvested. In subclass 1B, all plantations older than 21 years and 50 per cent of plantations in the age group 16-20 years are harvested. Thus scenario 1A represents a 'conservative forest management' option with rather long rotations, and 1B an 'aggressive management' alternative with very short rotations that promote an immediate supply of raw material from the plantations. In scenario 3A, all plantations older than 21 years are harvested, and in scenario 3B, all plantations older than 26 years and 50 per cent of plantations in the age group 21-25 years are harvested (similar to scenario 1A). Thus, 3A can be considered 'silvicultural optimum' scenario, while 3B represents a 'conservative forest management' scenario.

The results (Figure 5.11) show that in all cases (except scenario 2), the carbon pool of the radiata pine plantations increases with the maturity of the forest stands and increment of total plantation area. The 'aggressive management' scenario (1B) yields the

FIGURE 5.11
THE FIVE SCENARIOS FOR SIMULATING THE CARBON POOL IN RADIATA PINE PLANTATIONS BETWEEN 1995 AND 2015



lowest (130 Tg), while the scenarios for 'conservative management' (1A) and the 'silvicultural optimum' (3A) indicate the highest carbon pool at 175 and 173 Tg, respectively by the year 2015. The carbon pool in scenario 2 increases until the year 2005,

FIGURE 5.12A
TOTAL HARVESTED AREAS ACCORDING TO DIFFERENT SCENARIOS, 1995-2015

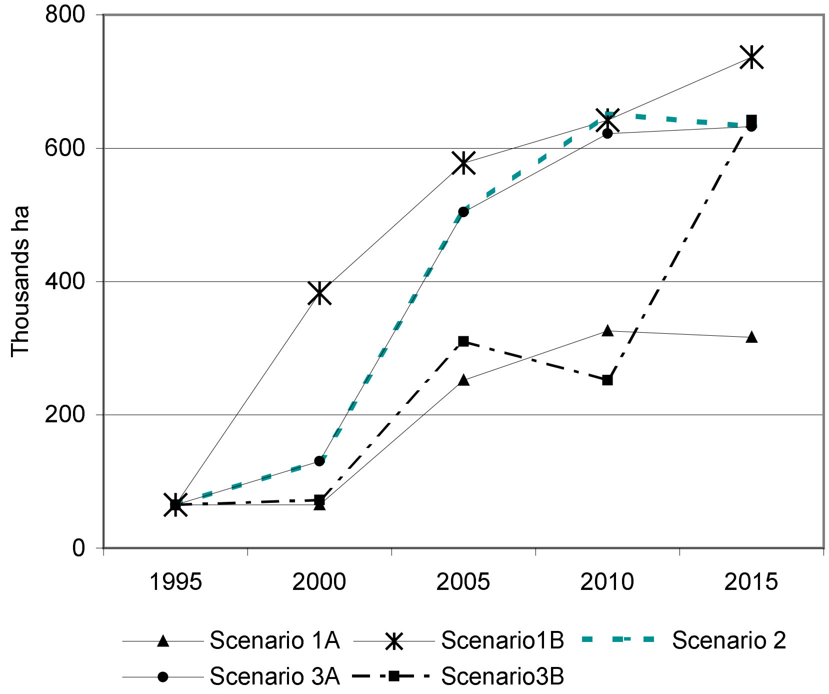
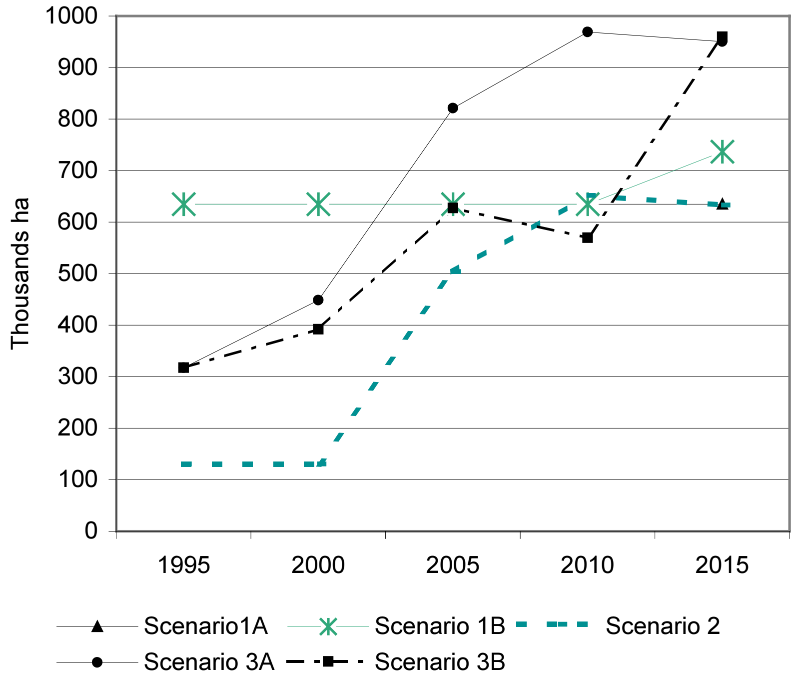


FIGURE 5.12B
PLANTED AREAS ACCORDING TO DIFFERENT SCENARIOS, 1995-2015



and decreases thereafter. This is explained by the fact that in scenario 2, only harvested areas are reforested and no afforestation is carried out. Consequently after 2005, plantations are dominated by young age groups, which have low carbon density per unit area. In scenario 1B, rotation is sub-optimal, and despite total plantation area increasing by 10 per cent, it is not sufficient to compensate the low carbon density of young plantations established after harvesting. The above scenarios also have differentiated consequences and impacts on the total plantation area, and areas harvested and planted annually (Figure 5.12A and 5.12B respectively). The 'silvicultural optimum' scenario, which is based on a net annual plantation area of approximately 32,000 ha in 1995, produces after the year 2006 the highest annual rate of plantation establishment (approximately 90,000 ha), yielding a 46 per cent increment in the total area of plantations by 2015.

4.2 Sustainable forestry and carbon policies

The development path followed by Chile has led to a two-faceted forest sector: on one side are the dynamic plantation-based forest industries, and on the other side, the backward perception of natural forests and the less-than-efficient industry based on these natural forests. Thus, it is a matter of debate whether this development strategy constitutes a sustainable approach. In recent years, development has been clearly unsustainable at least on the part of natural forests.

The above scenarios for radiata pine (Figure 5.11) show that plantation forestry—with its main objective of producing raw material for related industries—can play a diverse role in contributing to Chile's carbon balance. Under the 'silvicultural optimum' and 'conservative management' scenarios with relatively long rotation cycles, the carbon pool in 2015 is estimated to be approximately twice the 1995 level. Even under the 'aggressive' forest management scenarios with shorter rotations and large quantities of raw material being extracted for industrial use, the carbon pool increases by approximately 50 per cent.

It is also interesting to compare these scenarios in terms of areas planted annually and the total plantation area. High carbon pool values under the 'silvicultural optimum' scenario are obtained through the expansion of the plantation area by 46 per cent, whereas equally high pool values under 'conservative management' scenarios are obtained using long rotations and a 20 per cent increase in the plantation area. Low carbon pool values by 2015 are obtained when the current plantation area is maintained (scenario 2) or with 'aggressive' forest management approach with a 10 per cent increase in the plantation area.

The forest sector offers major opportunities for sustainable development. The first priority should be to adopt a more balanced approach in which the potential of plantation forestry, the rational management, and utilization of natural forests complement each other. A carbon policy for the mitigation of atmospheric CO_2 build-up needs to be integrated in the sustainable development strategy of the forest sector. This integration is indispensable because it is unrealistic to assume that forests will be managed solely for the purpose of carbon sequestration. On the contrary, it is vital that forests continue to be managed for their traditional economic benefits, but in a sustainable way (Apps and Price 1996). Sustainable forestry contributes to increasing the carbon sink effects, but also to halting significant net additions of carbon to the atmosphere that result from deforestation and mismanaged forestry practices.

Consequently, the criteria introduced by the Montreal Process (1995) for assessing sustainable forest management have to be taken into consideration in the formulation of forest policies which endeavour to promote carbon sequestration. Accordingly, due attention should be given to other issues like the conservation of biological diversity, maintenance of productive capacity of forest ecosystems, maintenance of forest ecosystem health and vitality, and the conservation and maintenance of soil and water resources. In addition, recognition must be given to issues such as the maintenance and enhancement of long-term multiple socioeconomic benefits to meet the needs of the society; legal, institutional, and economic framework for forest conservation, and sustainable management. The findings of the analysis on the limitations and rewards of sustainable forestry management in Chile could provide a solid foundation for dictating a new forest policy on a rational basis, and for launching new forestry programmes. A similar approach is recommended for other developing countries.

To enhance the maintenance of carbon pools in forest ecosystems and to increase the flux of CO_2 to forest biomass, several forest policy programmes should be adopted and implemented. The most important policy considerations are the promotion of plantations; the preservation of natural forests; the adoption of management regimes based on sustainable forestry; the rehabilitation of eroded land or degraded forests; the prevention of deforestation, and the production of industrial wood and fuelwood. The following relevant measures need to be included in such programmes (Brown *et al.* 1996a; Nilsson and Schopfhauser 1995; Kanninen 1993):

i) measures to prevent carbon emissions by conserving existing carbon pools in forest vegetation and soil through deforestation control, increasing the area of protected forests, changing harvesting regimes and the final utilization of wood, and controlling anthropogenic destructive actions like forest fires;

ii) measures to expand the storage of carbon in forest ecosystems by increasing the area and quantity of biomass and soil carbon density, and increasing carbon storage in durable wood products;

iii) measures to transfer forest biomass carbon into long-life products, substitute fossil fuel-based energy and substitute cement-based products with wood.

In Chile, of the three options mentioned above, only measures to expand carbon storage through the expansion of forest plantations have been partially adopted. Other elements, such as substituting fossil fuels with biomass energy, have future potential, but so far have not been considered. Natural forests and their management have been largely neglected and have, to some degree, been converted to other land uses, including plantations. The CO_2 emissions caused by this conversion and the low priority given to sustainable management of carbon pools in natural forests have only recently been off-set by the carbon sequestration of forest plantations.

4.3 Discussion

Without a doubt, the expansion of forest plantations, frequently over the last 20 years surpassing 100,000 ha of annually planted areas, has promoted the development of forest industries in Chile. The share of forest products of total exports of the last two decades has increased from 7.9 to 15.7 per cent (INFOR 1996). However, forest policies have mainly supported the establishment of plantations of fast growing exotic tree species to meet industrial needs. In contrast, promoting sustainable management and utilization of natural forests have been widely neglected. If one takes into consideration the high quantities of biomass and large carbon pools existing in these natural forests, this policy is inadequate from the point of view of carbon management.

Annual amount of carbon dioxide removed from the atmosphere by Chile's forest plantations, 49 Tg yr^{-1}, is substantial indeed and highlights their importance for carbon sequestration. Our scenarios show that both the carbon pool and carbon density will increase considerably in the future because biomass will increase as forest plantations mature. Currently, the forest plantations are characterized by a skewed age distribution in which young stands are dominant.

Data availability is the most important constraint in defining a method for estimating the carbon pools and fluxes in the Chilean forest plantations. In an ideal situation, estimations are conduced with detailed data on stand volume, increment and yield (distribution by age groups, site classifications, altitudinal variations, etc.). Currently, however, numerous GIS-based (geographical information system) methods are available, and this type of detailed spatial analysis can be implemented, but also accentuates the need for modern forest inventories to provide the basic stand and spatial data (Brown *et al.* 1996b). Unfortunately, as national-level forest inventory data are not available for Chile, detailed spatial information is also unobtainable. However, in the case of radiata pine plantations, application of the yield models by administrative regions at least partially takes latitudinal variations into consideration. In the case of eucalyptus plantations, data availability is even poorer and the volume estimation is carried out by using a simple growth model.

Monitoring changes in the carbon reservoirs is important for tracing the net flux of carbon between forests and the atmosphere. Forest inventories provide the best sources of data on tree volume and biomass because the data are generally collected in a statistically sound manner, but this also implies that national forest inventories are conducted at frequent intervals or on a permanent basis.

Soils are important reservoirs of carbon—their carbon content is often larger than in the terrestrial vegetation—and this fact should be taken into consideration in similar types of estimation (Kauppi 1996). Unfortunately, another limitation in the study is the paucity of data on the quantities of carbon fixed in forest soils. Several studies on this issue have been performed in other countries, but we considered it wiser to omit these calculations. The same criterion was applied in relation to avoid estimations on the amount of carbon stored in forest products, their average life, and the effect on the carbon balance of recycling material used in the industrial processes.

Future studies are needed to provide detailed estimations on the amount of carbon sequestered in forest plantations and also in natural forests. Natural forests, due to their larger standing volume and biomass, usually contain larger reservoirs of carbon than forest plantations. Thus, the conservation of natural forests has a positive effect on the national carbon balance. Currently, there are some 9 million hectares of national parks, 5.5 million hectares of forest reserves, and some minor areas declared as natural monuments. Although only a fraction of these protected areas is covered by closed forests, it is a clear fact that the policy of conservation has contributed to maintaining the existing carbon pools.

In Chile, plantations are in a key position in the formulation of carbon policies. Currently, only 28 per cent of the land considered suitable for forest growing is covered by different types of forests. At the moment, there is clear potential to increase forest resources through forest plantations, particularly by establishing plantations for rehabilitating degraded forest ecosystems, eroded and/or denuded lands, and for afforestation of marginal agricultural and grazing lands. Nevertheless, most of these areas are located in remote places where the creation of plantations is not economically feasible. In spite of their potential, new planting investments are constrained by land availability. However, if the current subsidies for establishing forest plantations are maintained, no significant reduction in the annually planted area is to be expected in the near future.

NOTES

[1] These include, among others, *E. globulus*, *E. nitens*, *E. delegatensis*, *E. fastigata*, *E. regnans*, and *E. camaldulesis.*

[2] The emission statistics were obtained from the internet page of the Energy Information Administration (EIA: http://www.eia.doe.gov/) and include emissions from fossil fuel consumption, from the consumption of petroleum, natural gas, coal, and the flaring of natural gas in 1996.

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CHAPTER 6

ECONOMICS OF TROPICAL FOREST LAND USE AND GLOBAL WARMING

Paula Horne

1. INTRODUCTION

The wide international concern over tropical forests and deforestation stems mainly from their global importance in supporting biodiversity and harbouring carbon stores. The prospect of global warming has given rise to demands for halting deforestation. Less attention, however, has been paid to understanding the underlying factors of deforestation. One of the prominent agents in the deforestation process is land-extensive farming, which accounts for over half of all deforestation (Myers 1993a). Extensive agriculture in forests or forest fringes provides livelihood to millions of families in tropical countries (Myers 1993b). The clearing and burning of forest land for agriculture contributes to global warming but on the other hand, provides subsistence for people.

This study presents a theoretical frame for considering the conversion of forests from the point of view of different interests. The main aim is to formulate an economically optimal decision-making policy on land use in relation to property rights, and functioning of markets. Section 1 discusses deforestation and its impact on carbon balance, and considers extensive cultivation and its alternatives in the face of the deforestation process. The underlying setting of decisions on land use is defined in section 2. Section 3 develops a model for analysing the different interests in forested land areas, specifying the reasons for divergence in the decision-making process. Finally, the model and its conclusions are discussed in section 4.

1.1 Carbon stocks and tropical forests

Tropical forests and deforestation have been one of the main issues in the discussion on global warming. Two concepts are essential in assessing the role of tropical forests in the carbon balance: carbon stock and carbon fluxes. The main stock of carbon, about 80 per cent, is stored in the oceans (Winjum, Dixon and Schroeder 1992). Of the terrestrial stock, above-ground vegetation stores about one-third, while twice as much has been estimated to be stored below ground in the soil, litter and peat (Dixon *et al*. 1994). Forests contain 80-90 per cent of all above-ground terrestrial carbon and 40 per cent of all below-ground carbon (Dixon *et al.* 1994). Tropical forests account for 50-60 per cent of the carbon stocks found in the world's forests (Myers 1993a). So, overall less than 10 per cent of the total carbon stocks are stored in tropical forests. However, with regard to carbon fluxes, the annual rate of deforestation in the tropics was 0.8 per cent over the 1980s (FAO 1995). With regard to the carbon balance, instability of the stock in tropical forests plays an important role.

Natural carbon fluxes of the marine and terrestrial systems are in balance; about 100 Gt are emitted in each system, and they both sequester about the same amount annually (Winjum, Dixon and Schroeder 1992). Man-made causes have resulted in an imbalance in carbon fluxes. In many tropical developing countries, the process of deforestation is the main emitter of greenhouse gases (Brown, Lugo and Chapman 1986; Faeth, Trexler and Page 1991; Houghton 1990; IPPC 1990) even though opposing views have also been presented (e.g. MacKenzie 1994). Changes in land use in tropical forest areas release 0.5-1.6 Gt of carbon per annum (Dixon *et al.* 1994; Houghton 1990; King *et al.* 1990).

The carbon balance depends not only on the quantity of carbon stocked in trees and soil, but also on the rate at which the carbon is released into the air. Burning forests releases stocks quickly into the atmosphere, but also produces other greenhouse gases such as methane and nitrous oxide (Marland 1988). Even when a burned or logged forest area is left to regenerate, total biomass declines drastically. As carbon constitutes roughly half of the total tree biomass (Dixon, Winjum and Schroeder 1993; Myers 1991), carbon stocks are likewise depleted. The biomass of a primary forest exceeds 600 t/ha, while that of a secondary forest is 50-220 t/ha. When an area is burned repetitively with no other vegetation except for grasslands ultimately remaining, biomass declines to the level of 8-10 t/ha (Peters and Neuenschwander 1988).

While tropical deforestation is seen at the international level as one of the culprits behind global warming, deforestation from the view point of a tropical country has two faces: socially desirable conversion of forests, and excess deforestation. The former concept could be called development instead of deforestation. From an economic point of view, land under forest cover is not utilized to its fullest, if an alternative land use brings more net benefits to the society in the long run. Excess deforestation results in actions which yield net costs to the society as a whole. In these terms, extensive or marginal cultivation is largely excess deforestation and will be looked at more closely in the following.

1.2 Role of extensive cultivation in tropical deforestation

Extensive cultivation (or shifting cultivation, swidden or slash-and-burn agriculture) is based on traditional methods and is still practised widely in developing countries. It is a land-extensive farming system with a high return to labour and is thus suitable for areas with low population density. There are regional and tribal differences in the method, but a few similarities prevail. The rotation cycle of slash-and-burn starts with the initial clearing of forested land (or shrubs) and burning of the biomass during a dry period. Different combinations of crops are planted before or after the burn. Cultivation is often done with basic tools powered by man. The importance of maintenance work such as weeding increases with the age of cultivation. A site is planted consecutively a few times and as yields diminish, the site is left to fallow for 7-10 years. A new plot is opened in another fallow site or in the forest. Traditional regimes of shifting cultivation were usually characterized by—and based on—low population densities averaging below 40/km^2 (Aiken and Leigh 1992). In the past, long fallow periods allowed the soil to recover and thereby ensured the sustainability of land use (Peters and Neuenschwander 1988).

In many countries, population growth among the slash-and-burn peasants is above the national average (Myers 1993a). With increasing population and changes in institutional structures, traditional shifting cultivation systems are no longer sustainable, while the lack of alternatives, scarcity of arable land, unequal land ownership systems, and demographic pressure continue to accelerate migration to marginal areas to practise extensive farming (Cornista, Javier and Escueta 1986; Hyde and Newman 1991; Kaul, Kanetkar and Achanta 1994; Kaul and Shah 1993). Knowledge of appropriate agricultural systems is often non-existent among migrant farmers and they use slash-and-burn methods on hilly sites causing erosion.

Extensive farming is one of the main causes of tropical deforestation. FAO's Forest Assessment Report (1982) cites extensive farming as the prominent reason for deforestation in Africa, tropical America and tropical Asia. It should be noted, however, that tropical countries are not a homogeneous group in this regard; case studies show that extensive agriculture cannot be blamed for the deforestation of India, for example, (Chakrabory 1994) or of Indonesia (Osgood 1994). Often extensive farming advances along logging roads into forests and the already degraded secondary forests are taken over for farming. While selective logging systems seldom deforest an area but does degrade it, extensive farming when practised in an unsustainable manner causes forests to be more or less permanently converted into other land uses.

1.3 Alternatives to extensive cultivation

There are two forest management options for controlling the balance of carbon fluxes. They can be classified as i) increasing of carbon sequestration through enhanced intensity and extensity of forests and ii) preserving the existent sequestered stock of carbon through forest conservation and halting deforestation.

An increase in carbon sequestration levels could be achieved through better management of existing forests and the expansion of forested areas. Estimations of the potential carbon sequestration possibilities, but yet unrealized by world forests, range from 1 to 3 Gt per annum for as long as a century (Dixon *et al.* 1994). Ensuring regrowth of logged-over areas and avoiding damage to the standing stock during logging operations would help intensify the long-term carbon stock in forestry sites. Expansion of forests to degraded lands through reforestation and afforestation would sequester substantial amounts of carbon (Dixon *et al.* 1994). Over 460 million hectares of plantations are needed to sequester an annual carbon increment of 3 Gt into the atmosphere. According to estimates, 1500-1800 million hectares of deforested land are technically suitable for forestation (Dixon, Winjum and Schroeder 1993), and plantation costs would rise well above US$ 180 billion, even with the opportunity cost of land excluded (Sedjo 1989). It is therefore not realistic to assume that forests would be conserved or planted merely for the benefit of carbon sequestration. The main priority of tropical forestry should be the more direct benefits with carbon sequestration as a positive side-effect.

In order to halt deforestation and to conserve existing stocks of carbon, the causes of deforestation should be examined and addressed. Mitigation measures should not only be effective, but also socially and economically acceptable. In many tropical countries, forest areas support substantial populations, a fact which needs to be acknowledged in any policy formulation. Mitigation measures will be feasible only if they serve the needs of local people involved in activities causing deforestation. In order to change the land-use practice of marginal farmers, the alternative offered must be socially, culturally, politically and economically acceptable. The methods in which subsistence is eked out from the marginal areas might be changed, but the people cannot be expelled from these areas in the foreseeable future. Despite efforts to regulate the conversion of tropical forests into extensive agriculture, it is an insistent problem in many tropical countries (see, for example, Cruz and Cruz 1990). As the tropical forest areas support large populations with limited alternative sources of income, the question of optimal forest areas has to be subjected to the needs of local peoples for any policy action to be effective.

Agroforestry systems with temporal or spatial intercropping of plants and trees could provide both food and income to local communities and contribute to carbon sequestration. One hectare of agroforestry, replacing 5-10 hectares of extensive cultivation, could sequester an average of 95 tons of carbon annually (Schroeder, Dixon and Winjum 1993). Hence, the contribution to carbon sequestration would be twofold; less pressure on the remaining forests and greater carbon storage in the agricultural systems. However, the direct costs of establishing an agroforestry are high, over US$ 450/ha on average (Schroeder, Dixon and Winjum 1993), and the yields can be harvested only after some time lag. Many farmers practising extensive cultivation have only limited resources on hand plus a high time preference. Hence, it is essential to look at the economics of land use in order to estimate the possibilities of transition.

2. THE FRAME OF LAND-USE DECISION-MAKING

Total value of the land is a subjective concept. International interest in tropical forests mainly concerns the benefits related to biodiversity and carbon sequestration; national interests lie perhaps in watershed management, tourism and logging, while the main concern of an individual farmer is the subsistence of his household. While the role of tropical forests in carbon dioxide sequestration may be conceived as a benefit by international interest groups, it is largely ignored by the individual farmer.

The individual farmer makes land-use decisions subject to an decision-making environment which includes different market, institutional and informational factors that regulate the costs and benefits from certain land use. Extensive work among migrant farmers would help to correct the lack of information on appropriate farming systems, but that is insufficient for a transition in farming systems to take place(DuBois 1990; UNESCO 1983). The use of unsustainable cultivation methods in extensive farming can often be attributed to market and institutional failures.

Imperfectly functioning or non-existent markets are a common feature of developing countries in general, and of environmental goods and services in particular. Many forest goods and services do not have markets, and many are of the nature of a public good. A manager who does not directly benefit from a common good would not be influenced by it in his land management decisions, even though the good might be of benefit to others. Such a good is said to be an externality in an individual's decision.

One prominent factor is the land tenure system (Barbier 1988; Southgate 1988; Southgate and Pearce 1988; Hyde and Newman 1991). Without land-use rights, a farmer will not invest resources in soil conservation, because the return is uncertain. A farmer with a more permanent tenure arrangement, for example, usufructuary rights or ownership, would consider the future land fertility important but would not be interested in whether cultivation causes off-site costs. Because of the institutional failure to assign property rights or to see to them being realized, the use of natural resources is exploitive. Socially suboptimal land use of a marginal farmer is a typical open access problem. Inadequate monitoring and control of land use have effectively rendered uplands an open access resource (Cruz and Cruz 1990). This is a well explored element in the literature of the economics of natural resources (e.g. Hartwick and Olewiler 1986; Neher 1990). Erosion control methods involve immediate investment in terms of labour —and at times capital—for gains that become apparent only in the future. With insecure land-use rights, the benefits from controlled soil erosion are disregarded in private decision-making. Any attempt to improve cultivation methods without addressing the question of land-use rights will not achieve long-term results (de los Angeles 1988; Barbier 1988; Horne 1996; Hyde and Newman 1991; Sawyer 1990; Southgate 1988; Southgate and Pearce 1988).

3. MODELLING LAND-USE DECISION-MAKING

3.1 Basic assumptions and definitions

This modelling of land-use decisions concentrates on the issues of property rights and environmental externalities. These issues have varying implications for different interest groups with varied aims over the use of tropical forest areas. The use of tropical forest areas is considered from the view point of four agents: global interest, social planner, sedentary farmer, and marginal farmer.

Global interest represents the general aim to conserve tropical forests. For simplification, global interest is represented only as the concern over the function of forests as a carbon sink and a biodiversity reserve. Therefore, assuming neither altruism nor demand for timber products, conversion of the forests is not desired. However, the global community has no property rights over forested areas in the tropics (Table 6.1).

Social planner represents national interest for the welfare of the entire nation. The optimal production of a social planner would equal that of aggregate private optima assuming neither market nor institutional failures (e.g. Varian 1992). A social planner has property rights and would internalize externalities because, for example, downstream effects of soil erosion would be just as much a matter of concern as the yields of upland cultivation. As global warming is by definition a global issue, it is not in the interest of a single country to make one-sided concessions to costly mitigation measures unless the directly accrued benefits in the country's own interest are expected to be higher than the costs. Hence, the benefits of carbon sequestration would be overlooked.

Sedentary farmer is defined here as one with property rights. Well-established usufractuary rights would result in the same effect; a farmer, being ensured of gaining future benefits, has the incentive to secure fertility of the land. Individual farmers have no concern of the externalities or of carbon sequestration.

Marginal farmer has no property rights, hence no incentive to retain fertility of the land. Again, the farmer has no regard for externalities.

TABLE 6.17
THE BASIC ASSUMPTIONS IN THE MODEL

	Property rights	Internalization of externalities	Internalization of sequestration benefits
Global interest	no	no	yes
Social planner	yes	yes	no
Sedentary farmer	yes	no	no
Marginal farmer	no	no	no

The accessibility of a forest site is approximated by transportation costs of the final agricultural goods. It is assumed that the market price of an agricultural produce reflects its home consumption value as well. The farmer is assumed to pay for the transportation of market goods, instead of devoting his own time to the task. This simplifies the handling of labour in the analysis. The payment is, naturally, related to the distance to market.

Output price is taken as an exogenous variable, reflecting an inelastic demand curve. This assumption is based on the low price elasticity of demand for food. In the case of a social resource planner, this is clearly an oversimplification, as he would be a price setter. However, the assumption is not expected to change the basic results of the model.

3.2 Modelling of optimal forest area

Given the assumptions, the global interest would be to maximize forest area. the social planner and the sedentary farmer strive to optimize the land area under agricultural production, and the marginal farmer would maximize agricultural production.

3.2.1 Global interest

Given that the global interest lies only in the conservation of forests as a carbon sink and a biodiversity reserve, no conversion of forest is desirable. The objective would be to maximize forest area, that is, to retain the present forest cover. There is no direct cost involved in conservation, thus the net benefit would equal gross benefit. The net benefit (B_G) can be written:

(1) $$B_G(t) = C[S(t)] + D[S(t)]$$

where:
- S(t) = stock of forested land and its soil resources
- C[S(t)] = benefit of carbon stock provided by S(t)
- D[S(t)] = benefit of biodiversity conservation provided by S(t)

Both benefits increase with the area of forest, $\partial C/\partial S > 0$ and $\partial D/\partial S > 0$. The objective is not time dependent.

3.2.2 Social planner

The social planner would optimize rather than maximize agricultural production, because he would also be concerned with, for example, the off-site erosion control of forest stock. Agricultural production function, Q(t) is given as:

(2) $$Q(t) = f[Z(t), S(t), X(t)]$$

where:
- Q(t) = agricultural production
- Z(t) = labour
- X(t) = distance to market

Available labour hours, Z, can be divided between agricultural tasks, Z_a, and soil conservation measures, Z_c, so that $Z = Z_a + Z_c$. The number of hours the farmer decides to devote to agricultural activities, Z_a, determines the level of agricultural production and is then the control variable. Both the output level and the level of environmental services are determined according to the farmer's choice, as shown below. As agricultural output and environmental services depend on the stock of forest, the stock of forested land is then the state variable. Effectively, the control variable, Z_a, determines the level of the state variable, *S.* Hence the dynamism of the model; the stock level is dependent on time through the past agricultural production. The objective function of a social planner can be written as:

$$(3) \qquad \max_{\left[Z_a(t)\right]} \int_0^\infty \{pQ(t) - wZ(t) - cX(t) + E[S(t)]\} e^{-rt} dt$$

subject to:

$$(4) \qquad S(T) = S(0) - a \int_0^T Q(T)$$

and:

$$(5) \qquad S(t) \geq 0$$

where:
- p = price of output
- w = price of input package
- c = cost of transportation
- E[S(t)] = benefit of erosion control provided by S(t)
- r = discount rate
- S(0) = initial stock of forest land
- S(T) = stock level at time T
- a = rate of soil erosion and nutrient depletion, function of ($Z_{a\,(+)}$, $Z_{c\,(-)}$)

The rate of change of *S(t)* is:

$$(6) \qquad \dot{S}(t) = -aQ(t)$$

The net benefits at time *t* of a social planner can be presented using a current value Hamiltonian (see e.g. Chiang 1992):

$$(7) \qquad \tilde{H}_S = pQ(t) - wZ(t) - cX(t) + E[S(t)] - u(t)aQ(t)$$

- u(t) = the product of the Lagrangian multiplier, λ, and e^{-rt}, shadow price of land.

There are two counteracting factors determining the optimal level of forested land stock; agricultural production and erosion control. The stock of forest and soil resources is reduced as production increases, therefore $[\partial S/\partial Q(t)] < 0$. As cultivation encroaches onto forested land, increased erosion reduces the productivity of land. This can be interpreted as an increase in the cost of agricultural production as resources are depleted, and a reduction in the future value of land. The decreasing on-site productivity of land due to erosion is an indirect cost of agricultural production. The shadow price of the land, $u(t)$, represents the foregone future benefits of agriculture, lost because of land degradation.

The choice of labour allocation, Z_a/Z_c, is reflected in the value of a. The constant a thus denotes the reduction of forest and soil stock resulting from agricultural activity. The higher the value of a, the higher the rate of land deterioration. The coefficient takes the highest value if all inputs are allocated to agricultural tasks. Investment in soil conservation measures would lower the value of a, but benefits would ensue after a time lag. This coefficient sets the inverse relation between the output and the stock, $(dS/dQ<0)$.

While the on-site effect of erosion is incorporated in $S(t)$, the off-site erosion along with other environmental services is included in $E(t)$. Ability of the land to control off-site erosion is an increasing function of the stock of forest land, $\partial E/\partial S(t) > 0$. As agricultural production is expanded at the expense of forested land stock, the environmental services are lost. Therefore, the foregone services are a direct cost of agricultural production.

Other costs of production include transportation and the cost of labour. The unit cost of labour, w, is its opportunity cost for example in non-farm employment. Assuming diminishing returns gives us $(\partial Q/\partial Z(t)) > 0$, and $(\partial^2 Q/\partial Z^2(t)) < 0$. Transportation cost is the unit cost, c, times distance, $X(t)$. The distance is an inverse function of the stock, $dX(t)/dS(t) < 0$. It is assumed that the distance is directly related to the total agricultural area, which, given the dynamic constraint above, is related to the stock. Constant transport costs per unit of output are assumed.

3.2.3 Sedentary farmer

The sedentary farmer differs from the social planner in that he places zero value to externalities, thereby ignoring the off-site erosion control. $E[S(t)]$ is thereby omitted from the objective function.

$$(8) \qquad \max_{\left[Z_a(t)\right]} \int_0^{\infty} \{pQ(t) - wZ(t) - cX(t)\} e^{-rt}\, dt$$

As the sedentary farmer has title to the land, he would take into account the impact of land degradation on the productivity of his own land, thus the stock constraint would hold, similarly to the previous case. The net benefits as presented through the current value Hamiltonian would be:

(9) $\tilde{H}_F = pQ(t) - wZ(t) - cX(t) - u(t)aQ(t)$

3.2.4 *Marginal farmer*

The farmer without land-title faces the same objective function as the sedentary farmer, but all labour is devoted to agricultural tasks. Consequently, soil deterioration would be more rapid. The farmer does not take the stock constraint into account; he maximizes the output and gains a net benefit:

(10) $B_L = pQ(t) - wZ(t) - cX(t)$

Optimal agricultural production would be initially higher than in the case of the sedentary farmer because all efforts are put into agriculture. Later, the higher rate of land degradation would require a larger area to be cultivated and the cost of transportation would increase.

3.2.5 *Rent-bid schedule*

Accessibility is one of the main factors controlling the process of deforestation. Several econometric studies have demonstrated the positive correlation between road density, total roads or proximity to markets and deforestation (e.g. Piña 1992). Opening up logging roads has facilitated the entry of farmers to previously inaccessible forest land. Apart from the physical difficulty of reaching a plot, lack of roads or long distance to market would increase transport cost, and cultivation would become less profitable. The rent-bid function explains the variance in rents of different sites in terms of their location. Holding other variables constant, rent becomes a function of distance. Rent from a marginal plot is zero, that is, marginal benefit equals marginal cost.

Using a rent-bid schedule we can show the different optimal stock levels diagrammatically. The rent-bid schedule is related to the total land area that has been cultivated by the time *t, aQ(t)*. Given a choice, the farmer locates in the proximity of markets to minimize transportation costs. The relation of transportation costs to the cultivated area over time is assumed to be linear to facilitate intuitive and diagrammatic interpretation. The distance from markets at the time *t, X(t)*, is then a function of *aQ(t)*.

Other components related to the agricultural area are the shadow price of land, and the off-site erosion control. The rent per unit of land from agriculture, R_i, can be written as:

(11) $R_i = pQ(t)_i - wZ_a(t)_i - \left[c + E[S(t)] + u(t)_i\right]\left[a_i Q(t)_i\right]$

where i denotes the decision maker.

For the purpose of drawing the diagram, the intercepts and slopes of the agricultural rents are identified in the following table.

TABLE 6.18
INTERCEPTS AND SLOPES OF THE AGRICULTURAL RENTS
IN THE BID-RENT SCHEDULE

	Intercept	Slope
Marginal farmer	$pQ_k(t) - wZ_{as}(t)$	$- c$
Sedentary farmer	$pQ_s(t) - wZ_{as}(t)$	$- c - u_s(t)$
Social planner	$pQ_m(t) - wZ_{am}(t)$	$- c - E - u_m(t)$

The benefit curve of global interest starts from the right hand side of the diagram presenting the rent from forest conservation. We can assume diminishing returns; the greater the supply of forested land, the less valuable the marginal unit of forested land, so the curve is downward sloping. With no cost for forest conservation, the preference would be to conserve all forests.

The intercept of the marginal net benefit curve is determined by the value of output and the labour cost. The marginal farmer would invest all labour to agricultural production and therefore the output is highest. If we release the assumptions that the output of a marginal farmer is also sold at markets at a price p, and that there is an opportunity cost to his labour, w, the basic results would not change.

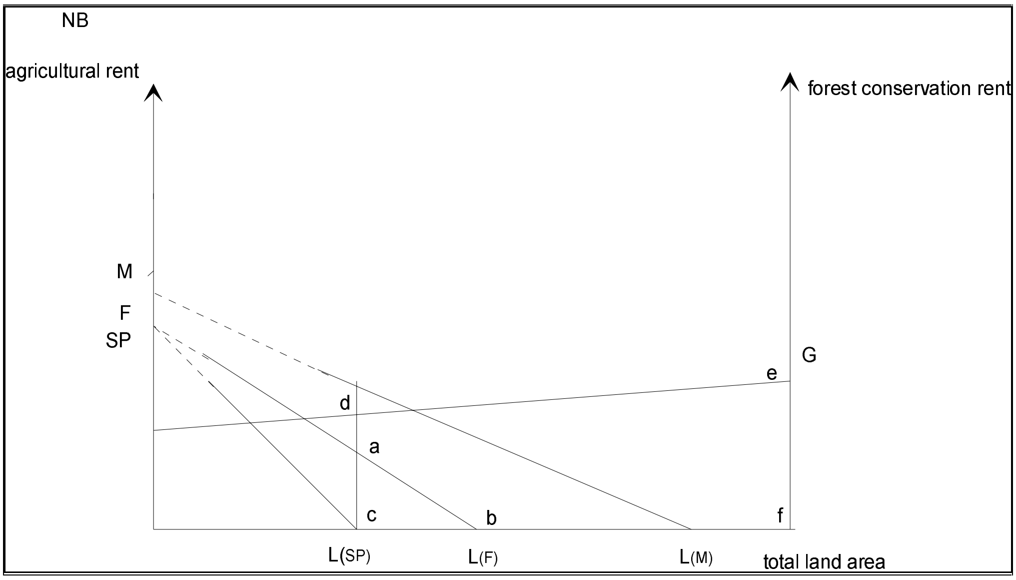
The slope of the curve is determined by three components; transportation cost, on-site erosion cost, and off-site erosion cost. Only the social resource manager takes account of them all. The sedentary farmer ignores the off-site erosion cost but takes the shadow price of land into consideration. The marginal farmer disregards all erosion costs, and the slope of his marginal benefit curve is flattest.

We would then have four different optimal options for forest cover. Figure 6.1 demonstrates the difference between the resultant forest covers. When marginal net benefit curves of agricultural production for the social planner (SP), the sedentary farmer (F) and the marginal farmer (M) cut the X-axis, the net benefit is zero, and the remaining (to the right) land area would be conserved under forest cover. The international interest in the maximum possible forest conservation is indicated by the benefit curve G. The dotted lines of the marginal net benefit curves are a reminder of the fact that a part of the previously cultivated land has deteriorated through soil depletion and erosion, if a is assumed to be positive for all farmers.

The marginal farmer, having no direct benefits from the forest, is interested in agricultural produce. If he had title to the land, like in the case of the sedentary farmer, he would be interested in sustaining the site's productivity and farming would be more intensive and less land extensive. Less forest land, from L(M) to L(F), would be converted for agricultural production.

It would be of national interest to conserve even a larger forest area because the forest provides environmental services such as soil conservation. If forest conversion was halted at the optimal level of the social planner, sedentary farmers should be given the incentive to reduce agricultural area from L(F) to L(SP) because they face a loss of revenue a-b-c. Nationally, supporting the transition from sedentary farming to an even less land-extensive agricultural system is preferred only up to the level indicated by the national optimum L(SP). The international community is a free rider enjoying the benefits of forests at no cost and with no property rights. At the level of national optimum L(SP), there would still be a benefit of c-d-e-f; consequently, it should be of global interest to subsidize the transition of farming practices.

FIGURE 6.19
THE NET MARGINAL BENEFIT CURVES OF LAND USE



4. DISCUSSION

The model presented in this study is a simplification of the decision-making environment related to land use in tropical forests. It shows that deforestation has as many facets as there are interest groups concerned with tropical forest lands. Deforestation will continue as long as it provides more benefits than costs to certain parties. Some of the simple assumptions made in the model may not seem realistic but do not change the basic results. Discount rate for calculating the present value of agricultural production is assumed to be the same for all parties, though in practise their time preference and accessible loan markets would differ substantially. However, changing the assumption would only make the argument stronger (Horne 1996). Similarly, the role of demographic pressure is not included in the model, but inclusion of the increasing demand for agricultural land would enlarge the areas converted.

Some deforestation could be in the interest of a country, if an alternative land use is socially more advantageous than keeping land under forest cover. Sustainable and productive agriculture is a prerequisite for fulfilling basic human needs and for relieving pressure on forest resources. The need for agricultural land is expected to increase considerably by the year 2020 (FAO 1995). The lack of capital for soil conservation measures and the increasing scarcity of arable land continue to put pressure on forest frontiers and areas opened up by logging.

It is widely recognized that trying to prohibit farmers from forest lands is practically impossible despite the fact that these people do not hold land titles (World Bank 1989). As has been seen, when people are struggling to meet their basic needs, the command-and-control method is an effective measure neither in practice nor theory. The collection of penalties from offenders is impractical when families live below the poverty line. If existing land-use practices are unsustainable by themselves, and produce a less than optimal forest cover, then alternatives that are socially, ecologically and economically acceptable should be supported by national policies. Community-based forest management or agroforestry systems are often adaptable to local conditions socially and ecologically, and new agricultural technologies generate crop varieties that provide higher yields, and more efficient cultivation methods. Poor farmers, however, often cannot afford the initial inputs needed, occupy land too poor or unsuitable for new methods (Aiken and Leigh 1992), or lack secure land tenures.

Correcting the existing institutional and market failures would bring land use closer to the socially optimal pattern. As governmental resources are often insufficient for effective management of forest lands, land reform would improve resource management. The need for land reform stems from actual open access situations currently under inadequate management and control. Firstly, it would allow the limited resources to be concentrated to a number of sites of special importance, ensuring proper control and monitoring of these areas. This involves the identification of the protection priorities, and the establishment of buffer zones. Secondly, both theory and practice have shown that resource management by private agents with established property rights is less destructive than under an open access situation. The sedentary farming system presented in this chapter would result in a higher level of forest conversion than is socially desirable. Transition to more intensive cultivation systems, e.g. agroforestry, might need initial support even though the incentive for adopting a land-conserving farming system should be in its financial profitability, not in external allowances (World Bank 1989). This support is not readily affordable in many countries.

The international community is presented here as a free-riding beneficiary of tropical forests existing as a carbon sink. To the extent that global warming is considered a costly problem and halting tropical deforestation considered the cure, economic support to the transition of land use seems to be a rational possibility. Transfer schemes, such as the joint implementation scheme, enable countries with high marginal abatement costs to invest in countries with lower marginal costs (Kaul, Kanetkar and Achanta 1994). The transfers would provide benefits to both the industrialized nations which can thus meet the required emission cuts without compromising the competitiveness of their industries, and to the tropical countries that are struggling with budget deficits. However, transfers would only be effective if they trickle down to change the farming systems causing deforestation.

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CHAPTER 7

POLICY OPTIONS IN CARBON SEQUESTRATION VIA SUSTAINABLE FOREST MANAGEMENT: AN EXAMPLE FROM THE NORTH

Hans Fredrik Hoen and Birger Solberg

1. INTRODUCTION

Biomass production on forest areas represents a main active sink in the global carbon cycle. The maximum level of biomass production is limited by area and available technology, e.g. tree species composition and different silvicultural management options like regeneration method, fertilization, release thinning in young growth or thinning. On a global level, we are far away from utilizing the maximum potential growth. Afforestation and intensified management of forested areas can increase the rate of carbon assimilation from the atmosphere. Over the last decades, the forests of several European countries have been a net sink of carbon (Kauppi, Mielikäinen and Kuusela 1992). An interesting question is whether this can continue in the future by changing forest management regimes or whether the absorptive capacity of these ecosystems is met.

Several types of models have been used to study this question, most of which have been pure simulation models.[1] Hoen and Solberg (1994) is the first (and to our knowledge, still the only) documented study of carbon sequestration in forestry using a dynamic optimization model. Compared with pure simulation models, optimization secures consistency in the evaluation of alternative forest management strategies. The model used in the study has in recent years been further developed regarding details and realism. The purpose of this chapter is to present an updated and more detailed analysis of potential and economic efficiency of carbon sequestration on a forested area, based on the framework presented by Hoen and Solberg (1994). Compared with Hoen and Solberg (1994) the analyses presented here are based on a far more detailed representation of the forest data, and more realistic modelling of regeneration and natural mortality. We have also included results when the flow of CO_2 out of and into the atmosphere is valued by a shadow price and is included in the cashflow of each timber management schedule.

The chapter is structured as follows; first we discuss some key methodological issues in general, then we present the new features of our model compared with the analysis reported in Hoen and Solberg (1994). This is followed by the results of the quantitative analysis along with discussion and conclusions based on the case study.

2. METHODOLOGY

2.1 Some general methodological issues

An analysis of the potential for increased carbon sequestration on forested areas can be divided into two fundamental parts; A) estimation or calculation of the physical flow of carbon (as carbon dioxide) between the atmosphere and forest biomass, and B) the evaluation, or ranking, of alternative forest management strategies producing different CO_2-flows through time. Problem A can be further subdivided into i) issues related to the estimation of forest biomass production such as growth rates, growth responses to different management treatments, growth responses to changes in the environment (for example climate changes, an issue not addressed in this chapter), regeneration probabilities and natural mortality, and ii) issues related to the estimation of how forest biomass decays over time after removal from the growing stand, where removal can be by natural mortality or by commercial harvests. The end use of the biomass is a key parameter in determining the time profile of biomass decay. Traditionally, pulpwood has been assumed to have a short 'life-cycle' resulting in complete decay after e.g. 2-5 years, while sawtimber obviously has much longer 'life-cycles' of maybe 50 to 100 years on average.

Problem B relates to the evaluation of alternative projects with different timing of benefits and costs through time. Different management strategies for a forest area might lead to quite different CO_2 flows over time (carbon sequestration profiles), and thus one would like to evaluate these to sort out the preferable strategies. See Price (1994, 1996) for a discussion of several aspects related to this issue. In addition, actions taken within forestry sector have to be evaluated together with actions taken in other sectors of the economy such as energy economizing or fossil fuel taxes. In this context it is important to clarify the particular features of forest biomass production in the global carbon cycle compared with other measures. We will point to three such features; tree biomass fixes large quantities of CO_2 for a long period of time, increments in the stock of living tree biomass provides added flexibility since the decision on the utilization of this biomass can be postponed for several decades in boreal ecosystems, and finally forest production represents a well-known technology, combined with well-functioning legislation and other forest policy traditions, in most industrialized countries. Based on this we think it is of particular interest from a climate change perspective to look at forestry actions with a medium-term time horizon of 30 to 50 years.

2.2 Model and assumptions

A case study is performed with a model for long-range forest management planning (LFMP) called GAYA-JLP (Hoen and Eid 1990). The quantitative analysis is thus based on calculation of forest-wide management strategies by simulating a number of alternative treatment schedules for each individual management unit in the forest and solving the forest management problems which are linked together by forest-level constraints, e.g. related to the harvesting profile over time, by means of linear programming (LP). The stand simulator GAYA is used to simulate alternative treatment schedules for each management unit (forest stand). As described in Hoen and Solberg (1994) a total tree biomass and carbon flow model is linked to GAYA. This model estimates the total biomass of a tree, divided into six categories: i) bark; ii) needles; ii) branches and stumps; iv) root system; v) sawtimber, and vi) pulpwood. When the removal constitutes a harvest, sawtimber and pulpwood are further divided into seven and two end uses, respectively. The carbon flow model converts the total biomass into CO_2 in tons and based on specific assumptions of decay within each end-use category, decay ratios and decay profiles after removal are calculated. The details of the assumptions can be found in appendix A of Hoen and Solberg (1994).

We applied the general LP formulation for a forest management problem consisting of k stands (calculation units):

(1) max $Z_p = \mathbf{c}^T\mathbf{x}$

subject to

(2) $\mathbf{A}_1\mathbf{x} \leq \mathbf{b}_1$

(3) $\mathbf{A}_2\mathbf{x} = \mathbf{b}_2$

(4) $\mathbf{x} \geq \mathbf{0}$

where:

Z_p : The objective function; in this example either NPV_{NOK}, NPV_{CO2} or NPV_{TOT} (see section 2.5 for definition) related to each stand treatment schedule.

$\mathbf{x}$: N x 1 (column) vector of decision variables, i.e. the number of hectares to be treated by a specific management schedule (activity).

$\mathbf{c}^T$: 1 x N (row) vector of known marginal net income related to each activity represented in **x**. The intertemporal dimension is taken care of as the marginal net income is represented as the net present value (NPV) of the sequence of net payments related to the activity.

$\mathbf{A}_1$: M x N (M rows and N columns) matrix of input-output production coefficients, e.g. the total harvested quantity, volume of sawtimber/pulpwood harvested, net payment or other state or flow variables related to the development on the forest area.

$\mathbf{b}_1$: M x 1 (column) vector of resource constraints (RHS values), one constraint related to each resource(row).

$\mathbf{A}_2$: K x N (K rows and N columns) matrix of area-related coefficients, which is 1 for element (i,j) if activity j belongs to timber class i and else 0.

$\mathbf{b}_2$: K x 1 (column) vector consisting of the initial area within each timber class.

Thus the analysis rests on the fundamental assumptions behind LP. With regard to forest biomass growth, the analysis can be regarded as piece-wise linear, since the area is divided into a number of different management units with individual productivity potentials expressed by state variables related to site index, tree species composition and density (number of stems per hectare), age and basal area. The valuation of timber harvests is based on the assumption of a fixed level for timber prices and harvesting costs irrespective of the total volume harvested. This may potentially be a serious problem when analysing larger areas, as one would expect an inverse relation between the quantity harvested and the marginal timber price within a certain period of time. In the case study presented here, this is not a major problem as the periodic harvest volumes during the planning period have been restricted to equal the average harvests during the 10-year period from 1983-92.

2.3 Data and simulation of stand treatment schedules

A case study is done for the productive forest area of a region in south-east Norway consisting of two counties, Buskerud and Vestfold. The area of productive forests is 695,000 hectares. At the time of inventory (1990/91) the growing stock was estimated to be $77*10^6$ m^3_{wb} (with bark), with 52 per cent in spruce, 32 per cent in pine and 16 per cent in broad-leafed trees. About 50 per cent of the growing stock is old-growth timber, covering 32 per cent of the area. The inventory consists of 2,350 observations, each representing a relascope sample plot. The plots were aggregated to 500 calculation units by using cluster analysis. A total of 10 clustering variables was used in the clustering procedure. The clustering variables represent the key explanatory variables for the growth model as well as the most important variables for calculating the logging costs. We have applied what we regard to be a medium level for timber prices and harvesting costs. The price for spruce is 400 NOK/m^3; 250 NOK/m^3 for sawtimber and pulpwood; 425 NOK/m^3 and 210 NOK/m^3 for pine correspondingly, and 290 NOK/m^3 as average for all assortments of broadleafed trees.

The calculations were done for a planning period of 30 years, divided into six 5-year periods. Thus, the simulated stand treatment schedules consist of different combinations of the available treatment options over the six 5-year periods. All treatments were assumed to take place in the middle of each period. Since the forest inventory registration was done in 1990-91, period 1 represents 1993, period 2 1998, and periods 3, 4, 5, and 6 represent the years 2003, 2008, 2013, and 2018 respectively.

Simulation of stand treatment schedules was done by allowing for 1 out of 7 available treatment options to be applied in each period. The management options were i) no treatment, i.e. continued growth (adjusted with natural mortality); ii) release thinning in young growth; iii) thinning; iv) fertilization; v) clearfelling; vi) clearfelling with retention of seed trees (pine) or shelter-wood cutting (spruce); and vii) planting or natural regeneration depending on the cutting regime. *A priori* feasibility requirements were defined for each of the management options except number i), in order to prevent the calculation of totally unrealistic treatment schedules. We have thus applied the same treatment options as in Hoen and Solberg (1994), but the definition of each treatment option and the feasibility requirements are more detailed in the current analysis. We have also conducted a sensitivity analysis with variable mortality rates depending on stand characteristics. Appendix 1 gives a detailed overview of the treatments and the feasibility requirements. A total enumeration of all combinations of allowed treatments, over six 5-year periods, for 500 calculation units resulted in between 10,500 and 11,500 stand treatment schedules for the different real rates of discount (RRD).

2.4 Modelling advances

The main advances of the modelling framework is related to a more detailed module for handling natural mortality within stands and far more detailed assumptions regarding regeneration of stands after clearfelling. Natural mortality can be given as the proportion of the number of living trees that will die each year, and this rate of mortality can be varied as a function of the state variables for a stand. Thus we can increase the mortality rate, e.g. for old, dense stands on highly productive land. There is a lack of empirically based mortality functions, but our approach allows for sensitivity analyses of what can be regarded as plausible subjective assumptions. In addition to this, we have included a check on the S%, i.e. the relation between dominant height and number of stems (Hart-Becking's spacing index) in the stand. If the S% falls below 10 on the best sites, natural mortality is increased in order to reduce the number of stems in the stand so that the S% is brought up to the 'high' target of 11. As the site index becomes poorer, the value of the low target on the S% is increased gradually up to 12, while the difference between the low and 'high' target is kept constant at one (1 per cent unit). The inclusion of an S%-induced natural mortality will lead to reduced estimates of timber production and biomass growth in stands with high initial density that is extensively managed (none or at most one thinning).

The assumptions related to regeneration are far more detailed compared to the previous study. First, the number of calculation units (management) is increased dramatically, leading to a larger variation in key variables as vegetation type, altitude and site index. The assumptions for regeneration are specified as a decision tree. An increased number of calculation units allows for more flexibility in the regeneration assumptions, i.e. more branches on the decision tree. Vegetation type is included as a variable for 'building' the decision tree. Based on subjective reasoning we have specified the tree species composition and density for the different alternatives of natural regeneration. The established tree number (sum of all species) varies from 1,000 to 5,500 stems per hectare, and the regeneration lag varies from 8 (5) to 30 (25) years when spruce (pine) is the main species. The costs related to natural regeneration varies from 0 to 1,500 NOK/hectare, and may include site preparation, weed control or modest supplementary planting. The assumptions for regeneration based on planting imply immediate replanting after clearfelling and the tree number of the main species varies between 2,000 and 1,600 with costs from 6,000 NOK/hectare to 4,000 NOK/hectare.

An important part of the timber growth projections in GAYA is related to the initial determination of the basal area when the dominant height of a stand is passing a certain preset limit. In the initial stand simulator, this was done simultaneously for all species (in mixed stands) when the height of the main species passed the preset limit. The actual height versus the preset height limit was checked once for each 5-year period. This is now changed so that basal area is initiated separately for each species and the actual height is checked year by year within the 5-year period. The initial approach gave a systematic over-prediction of growth levels for spruce and an under-prediction for pine and broadleaf trees. The systematic error was most profound in relation to spruce. In addition to this, the module estimating the initial basal area gave slightly erroneous estimates for stands with two or more species, leading to an overestimation of the total initial basal area of such stands. The growth model counteracted this error to some extent, due to the negative sign on the regression-parameter related to basal area mean diameter in the diameter increment functions. With the assumptions applied by Hoen and Solberg (1994) the errors related to initialization of basal area in mixed stands could lead to 5-10 per cent too high volume production over one rotation. In sum, the changes in assumptions, together with improvements and corrections of the stand simulator, lead to somewhat lower estimates of the timber production level over the entire rotation. Consequently this implies a reduction in the potential for carbon sequestration.

2.5 Forest management objectives

As mentioned the time horizon is set for 30 years. This means that forest management is allowed to vary within this 30-year period, while each stand is treated according to pre-specified silvicultural regimes beyond the 30-year time horizon. Each stand treatment schedule is thus continued until the preset rotation age is reached. For each stand, a standard rotation according to the pre-specified silvicultural regimes is simulated and the net present value of a perpetual series of this standard rotation in terms of timber and CO_2 is added. Thus, the valuation of the stand treatment schedules is done according to the Faustman tradition. These silvicultural regimes applied in the ending-inventory valuation are defined for each main species (spruce, pine or birch) and for different levels of the RRD. The regimes are set up to be close to a profit maximizing management regime where only timber is given a monetary value. The harvest level in each 5-year period was restricted to equal $3{,}640*10^6$ m^3_{wb} of sawtimber and $3{,}915*10^6$ m^3_{wb} of pulpwood. This represents the mean of the actual harvests of the logging seasons from 1983-92 based on statistics from the Central Bureau of Statistics (CBS). This constitutes 57 per cent of the current total increment and about 65 per cent of the harvest level in a forest management strategy satisfying a non-declining harvest flow constraint. We solved three LP-problems initially, the BASE problem with the net present value of the timber management cash flow (NPV_{NOK}) as the objective function, the CO_2 problem with the net present value of the flow CO_2 fixations (NPV_{CO2}) as the objective and the COMB problem with the net present value of the cash flow when both timber and CO_2 outputs are valued in monetary terms (NPV_{TOT}). The shadow price of CO_2 was set to 250 NOK/ton, which reflects the implicit shadow price of CO_2 emissions from a tax on gasoline of NOK 0.80 per litre. The CO_2-flow has a positive sign when CO_2 is taken from the atmosphere and transformed to organic carbon, and a negative sign when organic carbon decays and CO_2 is emitted back to the atmosphere. The difference in NPV_{CO2} between the CO_2 problem and the BASE problem give the total available increase in NPV_{CO2} by changing forest management practices during the six 5-year periods. Then we reformulated the BASE problem with NPV_{CO2} as an additional constraint (≥). Seven alternatives of this reformulated BASE problem were solved with a RHS value of NPV_{CO2} of 100 per cent,[2] 95 per cent, 80 per cent, 60 per cent, 40 per cent, 20 per cent and 0 per cent of the available increase added to the NPV_{CO2} level from the initial BASE problem. Two sensitivity analyses were made. In the first we examined the effects on NPV_{CO2} of prohibiting fertilization and planting and in the second we examined the effects of changes in the mortality rates for old-growth stands on NPV_{CO2}.

3. RESULTS

The production levels of NPV_{CO2} and NPV_{NOK} for the BASE, CO_2 and COMB problem are given in Table 7.1.

TABLE 7.1
PRODUCTION LEVEL OF NPV_{NOK} IN 10^9 NOK[2] AND NPV_{CO2} IN 10^6 TON
FOR THE BASE, CO_2 AND COMB PROBLEM

	BASE problem		CO_2 problem		COMB problem	
RRD	NPV_{NOK}	NPV_{CO2}	NPV_{NOK}	NPV_{CO2}	NPV_{TOT}	NPV_{CO2}
2%	17.1	37.0	14.4	52.4	28.7	50.2
3%	11.1	25.6	9.0	39.0	19.6	37.3
4%	8.2	19.7	6.4	31.1	14.9	29.7
5%	6.4	18.1	4.8	27.6	12.5	26.3

Note: The exchange rate for October 1999 is US$ 1 = 7.74 NOK.

From Table 7.1 we see that the difference in NPV_{CO2} between the BASE and CO_2 problem is $15.4*10^6$ ton CO_2 or 42 per cent of the level in the BASE problem when the RRD is 2 per cent. For RRDs of 3 per cent, 4 per cent and 5 per cent, the difference in NPV_{CO2} is 52 per cent, 57 per cent and 52 per cent of the level obtained in the respective BASE problems. The difference in NPV_{CO2} between the BASE and CO_2 problems can be split into two parts, namely the difference in CO_2-flow during the 30-year period when timber management is allowed to vary and the difference in CO_2-flow after the 30-year time horizon is passed. When splitting the NPV_{CO2} measure in two parts we find that 77.4 per cent is related to differences during the first 30 years when the RRD is 2 per cent. For RRDs of 3 per cent, 4 per cent and 5 per cent, the first 30 years contribute 78.2 per cent, 87.2 per cent and 90.2 per cent, respectively. We see that the results of the COMB problem are quite close to the CO_2 problem. This indicates that the shadow price on CO_2 sequestration (and emission) of 250 NOK/ton CO_2 is sufficiently high to dominate the solution. This is confirmed when we look at the marginal costs of increased NPV_{CO2}-levels in the BASE problem which are presented in Table 7.2. Even when 80 per cent of the available increase in NPV_{CO2} is obtained, the marginal cost is well below 250 NOK/ton CO_2.

TABLE 7.2
THE MARGINAL COSTS OF INCREASED NPV_{CO2}-LEVELS IN THE REFORMULATED BASE PROBLEM

	Real rate of discount (RRD) per annum			
RHS CO2	2% [3.09]	3% [2.68]	4% [2.26]	5% [1.91]
0%	0.2 (17.089)	0.0 (11.092)	0.1 (8.163)	0.1 (6.426)
+20%	20.7 (17.055)	26.6 (11.061)	26.4 (8.134)	25.1 (6.403)
+40%	51.2 (16.940)	47.5 (10.960)	52.5 (8.044)	53.9 (6.333)
+60%	90.7 (16.740)	81.6 (10.793)	82.9 (7.896)	83.4 (6.210)
+80%	201.8 (16.345)	186.5 (10.463)	166.1 (7.638)	153.4 (5.998)
+95%	713.9 (15.591)	581.0 (9.866)	607.8 (7.147)	641.5 (5.549)
99.99%	12058.9 (14.527)	13492.2 (9.087)	7090.4 (6.432)	9603.4 (4.851)

Note: The marginal costs, in NOK invested at time zero per ton increase in discounted net CO_2 fixation i.e. $d(NPV_{NOK})/d(NPV_{CO2})$, at increasing levels of the NPV_{CO2} constraint. RHS CO_2 gives the value of the NPV_{CO2} constraint, as the percentage of the maximum potential increase in NPV_{CO2} added to the NPV_{CO2} level obtained in the BASE problem. The figures in parentheses gives the value of NPV_{NOK} in 10^9 NOK, i.e. the objective function value in each LP-problem. The numbers in parentheses give the value of a 20 per cent fraction of the maximum potential increase in NPV_{CO2} in 10^6 tons, i.e. the difference between the BASE and CO_2 problem for the corresponding real rate of discount.

The marginal costs of CO_2 sequestration is increasing as the RHS-value of the NPV_{CO2}-constraint is tightened. The same pattern is found for all RRDs.

TABLE 7.3
THE AVERAGE AREA GIVEN DIFFERENT TREATMENTS IN THE BASE PROBLEM
WITH RRD AT 2 PER CENT PER ANNUM

RHS CO_2	Planting	Natural regeneration	Release thinning	Thinning	Fertilization	Seed tree	Clearfell
0%	7.9	24.3	23.3	3.4	22.5	1.4	26.2
20%	8.8	24.8	21.7	3.6	23.3	3.6	27.6
40%	9.7	25.7	20.9	4.0	25.4	7.7	29.3
60%	12.7	23.9	17.6	5.2	26.9	9.0	30.6
80%	16.6	23.6	14.1	5.4	28.7	12.0	34.2
95%	20.6	21.9	9.8	6.7	28.4	12.4	36.4
100%	25.8	29.8	5.9	7.3	28.5	10.1	39.5

Note: For release thinning in young growth, only the first five periods are included to avoid ending-inventory effects.

TABLE 7.4
THE AVERAGE AREA GIVEN DIFFERENT TREATMENTS IN THE BASE PROBLEM
WITH RRD AT 3 PER CENT PER ANNUM

RHS CO_2	Planting	Natural regeneration	Release thinning	Thinning	Fertilization	Seed tree	Clearfell
0%	6.8	26.7	17.4	2.7	17.6	2.4	27.4
20%	7.8	25.4	14.7	3.1	19.7	3.5	27.2
40%	9.5	25.3	13.8	4.2	22.7	6.5	28.7
60%	13.5	23.1	12.1	4.4	26.2	9.4	30.6
80%	16.6	22.1	8.0	5.6	28.4	11.9	32.6
95%	21.4	19.7	4.6	6.6	27.1	11.2	35.0
100%	26.7	16.4	3.5	8.6	27.3	10.3	37.0

Note: For release thinning in young growth, only the first five periods are included to avoid ending-inventory effects.

TABLE 7.5
THE AVERAGE AREA GIVEN DIFFERENT TREATMENTS IN THE BASE PROBLEM
WITH RRD AT 4 PER CENT PER ANNUM

RHS CO_2	Planting	Natural regeneration	Release thinning	Thinning	Fertilization	Seed tree	Clearfell
0%	6.1	26.7	13.2	2.7	15.3	2.8	26.8
20%	6.6	26.6	10.7	3.5	19.1	4.3	27.1
40%	7.1	28.1	9.0	4.1	22.7	7.4	29.1
60%	7.6	28.4	4.8	5.1	26.2	9.0	30.0
80%	12.9	25.1	2.7	6.3	27.6	10.8	31.9
95%	20.2	20.3	2.0	7.0	27.4	11.8	34.5
100%	26.0	16.7	1.9	7.6	28.1	9.3	36.6

Note: For release thinning in young growth, only the first five periods are included to avoid ending-inventory effects.

TABLE 7.6
THE AVERAGE AREA GIVEN DIFFERENT TREATMENTS IN THE BASE PROBLEM
WITH RRD AT 5 PER CENT PER ANNUM

RHS CO_2	Planting	Natural regeneration	Release thinning	Thinning	Fertilization	Seed tree	Clearfell
0%	6.5	26.7	8.9	1.8	11.7	2.6	27.2
20%	6.5	26.6	6.5	2.6	15.1	4.1	27.0
40%	7.0	25.7	2.8	4.0	19.4	5.1	26.6
60%	8.6	26.2	2.3	4.8	23.4	8.6	28.7
80%	12.9	24.1	1.3	6.1	24.9	10.7	30.9
95%	20.5	19.5	1.2	7.8	27.1	12.2	33.9
100%	28.4	13.5	1.6	8.1	27.6	8.7	35.9

Note: For release thinning in young growth, only the first five periods are included to avoid ending-inventory effects.

Tables 7.3 to 7.6 above present the changes in average area per 5-year period treated with different silvicultural treatments in the BASE problem with different RHS-values on the NPV_{CO2} constraint.

4. DISCUSSION

4.1 Changes between maximizing timber revenue and CO_2-sequestration

By inspecting the forest management prescriptions in the different LP-problems solved, we can trace the changes in forest management treatments. Three main changes seem evident; increased fertilization, reduced area treated with release thinning in young growth and increased planted area combined with increased area for final felling (seed tree/shelter-wood cutting and clearfelling). There is also a tendency towards increased thinning in the CO_2 problems compared with the BASE problems.

The area prescribed for thinning shows a large relative increase from the BASE problem to the CO_2 problem. The initial area, though, is fairly low. In the BASE problem thinning is mainly done in dense stands to avoid natural mortality caused by the S-% falling below the preset lower target as explained earlier. In many stands thinning is nonprofitable in terms of the NPV_{NOK} measure. Since the harvest volumes are fixed, it might be a good option in the CO_2 problems to thin dense stands and utilize the thinning removals for pulpwood and sawtimber instead of letting this volume be 'removed' from the growing stock as natural mortality. This also allows continued growth and biomass production in some other stands which are prescribed for clearfelling in the BASE problem. The effect of fertilization is straightforward since this treatment directly increases timber production (biomass growth) in an existing stand. The average area fertilized during the six 5-year periods is approximately 28,200 hectares in the CO_2 problem for all RRDs. Compared with the BASE problem, the fertilized area increased from 27 per cent with 2 per cent RRD to 55 per cent, 84 per cent and 136 per cent for RRDs of 3 per cent, 4 per cent and 5 per cent, respectively. During the last 10 years there has been virtually no tradition for fertilization of forest land among forest owners in the actual region. Table 7.7 gives the results from the same LP-problems as in Table 7.1, when fertilization is omitted as an treatment option.

TABLE 7.7
NPV_{NOK} IN 10^9 NOK AND NPV_{CO2} IN 10^6 TON FOR THE BASE, CO_2 AND COMB PROBLEM
NO FERTILIZATION

	BASE problem		CO_2 problem		COMB problem	
RRD	NPV_{NOK}	NPV_{CO2}	NPV_{NOK}	NPV_{CO2}	NPV_{TOT}	NPV_{CO2}
2%	16.9	35.2	14.3	50.4	28.1	48.2
3%	11.0	24.4	9.1	37.2	19.1	35.7
4%	8.1	18.9	6.4	29.4	14.5	28.2
5%	6.4	17.5	4.9	26.1	12.1	25.0

The NPV_{CO2} value in the CO_2 problem is decreased with 1,5-2,0*10^6 ton CO_2 when fertilization is omitted. The corresponding reduction in the BASE problem is 0.6-1.8*10^6 ton CO_2.

The average area treated with release thinning during the first five periods in the BASE problem is 23,300 hectares with 2 per cent RRD and 17,400 ha, 13,200 ha and 8,900 ha for 3 per cent, 4 per cent and 5 per cent RRD respectively. In the CO_2 problems, this is reduced with approximately 80 per cent. Release thinning reduces stand density and timber production. On the other hand, release thinning reduces the proportion of broadleaf trees and increases the size of the trees in the stand when clearfelled. This will be beneficial in terms of the CO_2-flow, since broadleaf trees produce only pulpwood (in the model) and larger trees have a higher sawtimber proportion with substantially longer time periods for decay compared with pulpwood. The area treated with release thinning in young growth in the CO_2 problem is nearly four times higher when the RRD is 2 per cent than when the RRD is 5 per cent.

TABLE 7.8

NPV_{NOK} IN 10^9 NOK AND NPV_{CO2} IN 10^6 TON FOR THE BASE, CO_2 AND COMB PROBLEM, NATURAL REGENERATION ONLY

	BASE problem		CO_2 problem		COMB problem	
RRD	NPV_{NOK}	NPV_{CO2}	NPV_{NOK}	NPV_{CO2}	NPV_{TOT}	NPV_{CO2}
2%	16.8	36.6	15.1	49.7	28.1	48.2
3%	10.9	24.8	9.6	36.0	19.0	35.1
4%	8.0	19.7	6.9	28.8	14.5	28.1
5%	6.3	18.1	5.4	25.5	12.1	24.8

The average area planted per 5-year period is between 6,100 and 7,000 ha and the average area clearfelled is between 24,300 and 26,700 ha for all RRDs in the BASE problem. The constraint on timber harvesting volumes limits the area that can be clearfelled and regenerated. This explains the nearly constant level for all RRDs in the BASE problem. In the CO_2 problem, the area clearfelled is increased by 51, 35, 37 and 32 per cent for RRDs of 2, 3, 4 and 5 per cent. The corresponding increase in planted area is 227, 293, 326 and 337 per cent. The gain in terms of CO_2 sequestration may be caused by several effects. First, in the CO_2 problem, planting is preferred to natural regeneration since more dense stands are established and establishment is done faster. Secondly, we might have an ending-inventory effect related to regeneration. In the management programmes for calculating the ending-inventory value, natural regeneration is assumed as the regeneration method on increasingly higher sites as the RRD increases. The LP-model will evaluate the gain of concentrating clearfelling to lower sites during the simulation period (six 5-year periods) to avoid having these stands regenerated naturally in the ending-inventory evaluation. Stands on the best sites will be regenerated with planting in the ending-inventory evaluation, so it might not be worthwhile to clearfell and regenerate these stands, given the limitation on clearfelling area caused by the constraints on harvesting volumes during the 30-year period. We have not yet tried to isolate the magnitude of this potential effect. Table 7.8 provides the results of a sensitivity analysis on NPV_{NOK} and NPV_{CO2}, when planting is omitted as an available method for regeneration during the six 5-year periods.

If we compare the effects achieved by omitting planting with the effects of omitting fertilization, i.e. Table 7.7 versus Table 7.8, we see that planting seems slightly more important than fertilization for increasing CO_2 sequestration.

By comparing the changes in forest management when 80 per cent and 100 per cent of the available NPV_{CO2} increase is forced on the BASE problem, it seems that the most costly actions are related to changes in clearfelling and regeneration priorities. Since these might be due to ending-inventory effects, they cannot be regarded as very realistic estimates.

The restrictions on harvest level implies that the volume of removal (harvested volume plus volume of naturally died trees) is significantly lower than the current growth. This implies that the volume of the growing stock is increasing over the 30-year period. The average age of the stands (area-weighted) also increases over the 30-year period. Assumptions related to natural mortality are uncertain, and we believe that this uncertainty increases with the age of the stands. We applied the following assumptions for natural mortality to check the sensitivity related to the NPV_{CO2}-measure. The general mortality rate was increased from 0.4 per cent to 0.76 per cent of the trees per year. When stands with a tree number above 500 stems per hectare passed a certain age, the annual mortality rate was doubled to 1.52 per cent. The relative diameter of timber dying naturally is assumed to be 0.7 in all cases. This age limit was varied with site index, which was divided into three classes. The age limit was 80 years for H_{40}>18.5, 100 years for 18.5>H_{40}>12.5 and 120 years for 12.5>H_{40}. Table 7.9 gives results from this analysis.

TABLE 7.9
NPV_{NOK} IN 10^9 NOK AND NPV_{CO2} IN 10^6 TON FOR THE BASE, CO_2 AND COMB PROBLEM, INCREASED NATURAL MORTALITY

	BASE problem		CO_2 problem		COMB problem	
RRD	NPV_{NOK}	NPV_{CO2}	NPV_{NOK}	NPV_{CO2}	NPV_{TOT}	NPV_{CO2}
2%	15.9	33.1	13.5	49.2	26.9	47.0
3%	10.5	22.7	8.5	36.4	18.4	34.8
4%	7.8	17.7	6.0	28.8	14.0	27.3
5%	6.2	15.9	4.6	25.5	11.7	24.1

The results given Table 7.9 show that these changes in mortality assumptions cause an effect on NPV_{CO2} in the CO_2 problem of roughly the same magnitude as omitting fertilization or planting. It should be mentioned that a 30-year period is a short time for achieving the full effect of these assumptions, since rotation ages in the ending-inventory evaluation exceed the ages for introducing the high level of mortality (1.52 per cent of the tree number per year) only on the poorest sites.

4.2 Changes in relation to previous study

The results obtained in this analysis indicate a much larger difference in NPV_{CO2} between the BASE and CO_2 problem than what was obtained in Hoen and Solberg (1994). There are several explanations for this. First, the increased variation in the assumptions regarding regeneration will lead to a greater variety in the estimates of forest biomass production. Second, the increased number of calculation units in this study gives more flexibility for the solution of the LP model and thus one would expect a larger difference between the two solutions. Looking at the aggregated value of NPV_{CO2} for the different RRDs, this analysis provides lower estimates per hectare compared to Hoen and Solberg. The reduction is in the range of 18-26 per cent. This can be explained partly by the variation in input data and partly by the improvements and corrections of the stand simulator. In the county of Vestfold which is included in this study, but not in the previous study, nearly 31 per cent of the growing stock is broadleaf trees, which in this case is assumed to produce only pulpwood. In Buskerud 14 per cent of the growing stock is broadleaf trees, indicating that a much larger proportion of the growing stock can be used as sawtimber. Also, the pulpwood ratio of the total harvest during the 30 year period is 0.52 in this analysis, while in Hoen and Solberg (1994) the pulpwood ratio was 0.47. One could expect the NPV_{CO2} measure to be fairly sensitive to a (permanent) change in the mix between sawtimber and pulpwood. We tried to check this by changing the mix of sawtimber and pulpwood for the six 5-year periods and resolved the BASE, CO_2 and COMB problems. Out of the total harvest of $7.555*10^6$ m^3_{wb} per 5-year period, 40 per cent was restricted to be pulpwood and 60 per cent sawtimber. Results of this sensitivity analysis are given in Table 7.10.

TABLE 7.10
NPV_{NOK} IN 10^9 NOK AND NPV_{CO2} IN 10^6 TON FOR THE BASE, CO_2 AND COMB PROBLEM, 60% SAWTIMBER AND 40% PULPWOOD IN TOTAL HARVEST

	BASE problem		CO_2 problem		COMB problem	
RRD	NPV_{NOK}	NPV_{CO2}	NPV_{NOK}	NPV_{CO2}	NPV_{TOT}	NPV_{CO2}
2%	17.3	37.6	14.8	52.1	28.8	49.9
3%	11.4	27.0	9.5	39.1	19.9	37.4
4%	8.5	21.1	6.9	31.2	15.3	29.7
5%	6.8	19.2	5.4	27.9	12.9	26.7

By comparing Table 7.5 with Table 7.1, we see that the effect of changes in the ratio of sawtimber/pulpwood does not alter the level of the NPV_{CO2} parameter very much in the CO_2 problem. In the BASE problem NPV_{CO2} is somewhat higher with a higher sawtimber ratio, and the difference is increasing in relative terms when the RRD increases. It is however also likely that the ending-inventory valuation might produce unexpected and counterintuitive results in this case.

The results in this study clearly demonstrate that there are significant differences in carbon sequestration between a forest management strategy based on maximization of discounted timber profits (NPV_{NOK}) and maximization of discounted net CO_2 sequestration (NPV_{CO2}) for the actual region. The strategy based on the maximization of the discounted cash-flow when both timber and CO_2 sequestration are valued in monetary terms is very close to the pure CO_2 strategy for the NPV_{CO2} measure, while the sacrifice related to pure timber profits (NPV_{NOK}) is modest. This result underpins the fact that the marginal cost of NPV_{CO2} is increasing sharply for the last 10-15 per cent of the potential increase in this factor. The difference in NPV_{CO2} between the BASE problem without fertilization and the CO_2 problem with fertilization is $17.2*10^6$, $14.6*10^6$, $12.2*10^6$ and $10.1*10^6$ ton CO_2 with RRDs of 2, 3, 4 and 5 per cent per annum. By taking 80 per cent of this increase and calculating the 30-year annuity, we get 0.614, 0.596, 0.564 and 0.778 million ton CO_2 as the yearly increase in net CO_2 sequestration for the RRDs of 2 per cent, 3 per cent, 4 per cent and 5 per cent respectively.

The results from the analyses undertaken in this chapter confirm the findings presented in Hoen and Solberg (1994) that also in boreal forest one can get considerable increase in carbon sequestration by changing forest management regimes. Furthermore, the more detailed data and assumptions for timber management and especially for regeneration applied in this study gave much higher differences in NPV_{CO2} between the BASE and the CO_2 problem compared with the first analyses. By valuing the CO_2-flow for each timber management schedule with a shadow price of 250 NOK/ton CO_2 and maximizing the NPV_{TOT}, this provides solutions very close to the CO_2 problems (pure NPV_{CO2} maximizing problems).

APPENDIX

A presentation of how the stand treatment management options are defined and what requirements a stand must meet in order to be allowed to adopt the actual treatment, is given in this appendix. Treatments and feasibility requirements were defined in order to reflect the present forest management practices among forest owners in the counties of Buskerud and Vestfold. In addition, the flexibility given in the feasibility requirements assures the simulation of a number of 'unusual' treatments. The following treatments were considered possible in each period:

1. No treatment. The maximum life-time of a stand was set sufficiently high to allow all stands to be submitted to this treatment in all six 5-year periods.

2. Release thinning in young growth, the regulated tree number was set to 1,900 trees/ha, 1,700 trees/ha and 1,500 trees/ha for the site classes (H_{40}) above F18.5, between F15.5 and F18.5 and below F15.5 respectively. The allowable top height range was set from 2 to 6m.

3. Thinning, removal of 30 per cent of the basal area with a 0.85 ratio between the diameter in the removal and the initial stand. The minimum initial basal area was set to $20m^2$/ha. The top height of the stand was to be between 10 and 18m, and thinning could only take place up to two times during one rotation.

4. Fertilization, 500 kg of ammonium nitrate which is equivalent to 172.5 kg of nitrogen, for sites between G11 and G20 with a total age between 45 and 70 years. Fertilization could only take place twice during one rotation, with a minimum of two period lags, i.e. 10 years, between the operations.

5. Clearfelling, the minimum and maximum age was set at 60 and 110 years for site classes above F15.5, 70 and 150 years for site classes between F9.5 to F15.4 and 100 and 170 years for site classes below F9.4.

6. Clearfelling with retention of seed trees in pine for all site classes. The minimum top height for this treatment was set to correspond with the minimum ages for clearfelling used for pine. Shelterwood cutting in order to stimulate natural regeneration in spruce was available for two vegetation types covering approximately 20 per cent of the spruce area. The top height interval was set to 18-30m and the interval for total age was set at 50-180 years.

7. Planting or natural regeneration depending on the cutting regime, i.e. clear cutting or regeneration harvesting.

NOTES

[1] Cf., for example, Apps and Price (1996) or Birdsey and Heath (1995).

[2] The RHS-level was set at 99.99 per cent to avoid the extreme shadow price that would be obtained if the RHS-level were set at 100 per cent (no flexibility).

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CHAPTER 8

COST EFFICIENCY, ENVIRONMENTAL IMPACTS AND POTENTIAL OF CARBON SEQUESTRATION IN FORESTRY IN INDONESIA

Birger Solberg and Boen Poernama

1. INTRODUCTION

Possible global climate changes due to increased concentration of climate gasses are one severe challenge facing humanity today. Changes in forest management may be one important factor in reducing the concentration of CO_2 in the atmosphere. The potential of CO_2 sequestration in forestry varies greatly between different climate regions, with the tropical region having the highest growth potential per area. Indonesia is one of the most important tropical forestry countries in the world. However, only a few studies have previously been done on carbon sequestration for this country (Brotoisworo 1991; Makundi *et al.* 1992; Murdiyarso 1991; Suwanda 1990). A common feature of these studies is that they do not give any economic analysis of the costs and benefits of various sequestration options.

The main objective of this chapter is to analyse the cost efficiencies, environmental soundness, and carbon sequestration potential of some forest management strategies in Indonesia.

A short study such as this can only give the main assumptions and results. For a more detailed description, the reader is referred to Anon (1994), Lillethun (1994a, b), Solberg and Purnama (1994), and Sørensen (1994). Although based on these studies, this study gives a deeper and more thorough analysis of their combined results.

2. METHODOLOGY AND MAIN ASSUMPTIONS

2.1 Forest management strategies

The following management strategies are analysed (G refers to strategies on land which is grassland; P refers to strategies in lowland evergreen rainforests allocated to production forests having less than 20 m^3 per ha of standing volume of commercial species with dbh greater than 30 cm):[1]

G Management strategies for Imparate grassland (*alang-alang* land)

G1 Secondary forest establishment (by introducing fire control, natural regeneration, some enrichment planting). After 60 years it is assumed that selective felling are practised as the TPTI system with 35 years between fellings;

G2 Monoculture plantation, planting *Paraserianthes falcataria*, 10 years rotation;

G3 Monoculture plantation, planting *Pinus merkusii*, 30 years rotation;

G4 Mixed plantations, planting *Paraserianthes falcataria, Acacia magnus,* and *Eucalyptus spp.*, 10 years rotation;

G5 Monoculture plantation, planting *Shorea spp*. with shading trees in the initial stage, 60 years rotation;

G6 Monoculture plantation, planting *Tectona grandis*, 60 years rotation;

G7 Permanent grassland. No actions. This is the base-scenario for this group of forest management strategies;

P Management strategies for lowland evergreen rainforest land allocated to production forests having less than 20 m^3 per ha of standing volume of commercial species with dbh greater than 30 cm[2]

P1 Selective felling (TPTI);

P2 Clearfelling of rainforests, then G2;

P3 Clearfelling of rainforest, then G3;

P4 Clearfelling of rainforest, then G4;

P5 Clearfelling of rainforest, then G5;

P6 Clearfelling of rainforest, then G6;

P7 No harvest—leaves the forest as is. This is the base-scenario for this group of forest management strategies;

2.2 CO_2 sequestration

For each forest management strategy, the net fixation of CO_2 in year t, R_t, is calculated as:

$$(1) \qquad R_t = G_t - \sum_{i=1}^{N} D_{t;i}$$

(2) $G_t = v_t \cdot s \cdot f \cdot k$

where

G_t is the gross fixation of CO_2 in forest biomass at time t

$D_{t,i}$ is the decay of CO_2 at time t from forest biomass component i

N is the total number of biomass components considered

v_t is the volume growth of forest in year

s is the specific weight of the species analysed

f is the fraction of carbon in dry wood (for nearly all species equalling 0.5)

k is the fraction between the molecular weight of CO_2 and atomic weight of carbon, i.e. 44/12

(3) $D_{t,i} = A_i \cdot k \cdot q_i \, (1 - q_i)^{t-p}$

where

A_i is the carbon content of the biomass component i starting to decay at time p ($p<t$)

p is the time when the biomass component i starts decaying

$D_{t,i}$ is as defined in (1)

q_i is the percentage decay per year of the biomass component i

(4) $q_i = 1 - 0.1^{\frac{1}{n}}$

where

n is the time (in years) from when the biomass component starts decaying until 90 per cent of the biomass has decayed

q_i and i are as defined in (1) and (3).

The biomass and carbon content have been estimated based on Soerianegara (1965), Soemarna 1992 a, b), Soewarsono (1990) and Suharlan *et al.* (1975).

2.3 Cost efficiency

Cost efficiency is used to compare alternative ways of reducing atmospheric carbon and to rank these alternatives according to their cost efficiency in reducing the concentration of atmospheric CO_2, as we cannot estimate the economic benefits of such a reduction, only the physical quantity and the costs. Somewhat imprecisely, we estimate the 'environmental benefit per net invested dollar', where the environmental benefit is the net fixation of atmospheric CO_2 measured in physical terms (tons). More precisely, cost efficiency is calculated as

$$(5) \qquad E = \frac{M}{K}$$

where

$$(6) \qquad M = \sum_{t=0}^{T} R_t (1-p)^t (1+r)^{-t}$$

$$(7) \qquad K = \sum_{t=0}^{T} (C_t - U_t)(1+r)^{-t}$$

Here, R_t is the fixation (or emission) of atmospheric CO_2 (in tons per ha per year) at time t, r is the discount rate, p is the relative reduced value of further reduction of emissions[3], C_t is gross costs (negative numbers) of the project implementation at time t and U_t is the gross income at time t. T is the length of period the analysis covers; in this report, T is infinite as we have assumed an infinite number of forest rotation periods (including the end use of the wood products) for each alternative analysed. C_t and U_t are specified in Solberg and Purnama (1994).

The difference between C_t and U_t may be either positive or negative. If a project is profitable in itself without consideration for the positive CO_2 effects, we have a so-called 'no-regret' alternative.

Negative R_t indicates a net emission of CO_2 to the atmosphere. For forest plantations we usually get a positive R_t the first 10-100 years, and then a negative R reflecting the decay of the wood fibre and a corresponding net emission of CO_2. T is here the infinite horizon.

Based on Equation (5) - (7) the various measures (projects) to reduce the concentration of atmospheric CO_2, can be classified in three groups:

i) The first group of projects is characterized by a positive M and a negative K. These are projects which are profitable to implement *even without* consideration for the CO_2 effect. As such, they are 'no regret' options with respect to reducing the concentration of atmospheric CO_2.

ii) The second group consists of measures which have a positive M and a positive K (i.e. the costs C are larger than the monetary benefits U, discounted). Here, Equation (1) gives a certain number: for example 5, indicating that we have a cost efficiency of 5 kg CO_2 per $ net investment. The higher the figure, the more cost efficient is the project relative to other projects.

iii) The third group of projects has negative M, meaning that the discounted net emission is higher than the fixation. This can happen if the project, for example, negatively influences the concentration of carbon in the soil, or as for strategy P1 where there is a negative emission of CO_2 over time. Such projects are of rather limited interest with regard to sequestering atmospheric CO_2.

The choice of calculation rate of interest is decisive for which of the groups i) or ii) a project falls into. In this analysis we have used 7, 10, and 13 per cent per annum real rate of interest. In Norway, a rate of interest at 7 per cent per annum is recommended by the Ministry of Finance in the evaluation of public projects, and the Norwegian analysis on cost efficiency of carbon sequestration in forest biomass (cf. Solberg and Hoen 1996) is based on this figure. One may argue that a higher interest rate is justified for Indonesia, (because, among other things, of probable higher needs for increased consumption today to cover basic needs as compared to Norway). Therefore, the analyses in this report have been calculated at the 10 per cent and 13 per cent per annum rate of interest. It should be mentioned, however, that a real (and risk free) rate of interest in the range 7-13 per cent per annum is high.

Section 3 shows that only management strategies G1 and G3 fall in the above mentioned category ii). All the other management strategies analysed are so profitable that they fall in category i), i.e., they are no regret options.

The end-use of forest products and decaying of wood residues are included in the analyses of the net carbon emissions. However, it should be emphasized that the possibilities of burning wood products, thus providing a substitute for fossil fuels after their anthropogenic use, is *not* included in the analysis. To the extent that such a substitution is possible in Indonesia, the analysis will therefore underestimate the value of forests as carbon sink.

The following foreign exchange rates have been used:

1970 Rps per US$ for 1991

2000 Rps per US$ for 1992

2080 Rps per US$ for 1993.

Costs, revenues, wood end-uses, decay times of wood and forest products, tree planting growth, tree-specific weights, etc. are as stated in Solberg and Purnama (1994). All utilized economic data are in real term 1991 prices. Alternative land costs are valued at zero, as it is assumed that the marginal grassland considered here gives close to zero economic benefits, and that areas currently as forest land cannot by law be transferred to other land uses.

2.4 Environmental impacts and area potential

The environmental impacts (except carbon sequestration) of the forest management strategies defined in section 2, were assessed as described in Lillethun (1994b). Nine criteria were used:

- i) Occurrence and stability of a tree layer
- ii) Canopy structure and height in mature stands
- iii) Species composition of tree layer
- iv) Habitat for other vegetation and valuable fauna/effect on biodiversity
- v) Protection function with regard to erosion
- vi) Hydrology; surface water flow, soil moisture, and groundwater recharge
- vii) Soil properties, nutrient recycling (import/export of nutrients/carbon) and sustainability of timber production
- viii) Effect on local climate
- ix) Actual and probable use of chemicals.

For each of these criteria, a scoring scale of 6 (1 lowest, 6 highest) was used, and the total score was summed linearly. The scoring was set in cooperation with the Biodiversity Working Group of Indonesia.

Based on existing statistics from various sources, the land potential for the various forest management strategies was estimated as described in Sørensen (1994), and as shown in Table 8.2. The maximum and minimum estimates refer to what is assumed to be realistic implementation of the various forest management strategies, respectively 40 per cent and 10 per cent of the total available area.

3. RESULTS

Table 8.1 shows the overall ranking with regard to environmental soundness. Table 8.2 gives the potential areas available. Here, compared to Sørensen (1994), we have reduced the area available for teak by 50 per cent, because we believe this species is more site-quality demanding than other species considered.

Table 8.3 gives the estimated carbon potential sequestration, economic preferability, and environmental soundness of the forest management strategies considered to be acceptable (neither P2 nor P3 are considered acceptable from an environmental soundness point of view). When going from the overall ranking in Table 8.1 to the three-categorized ranking of environmental soundness in Table 8.3 (last column), low means below 30 points and high means above 40 points in Table 8.3.

TABLE 8.1
OVERALL RANKING OF ENVIRONMENTAL SOUNDNESS OF MANAGEMENT STRATEGIES BASED ON SEPARATE EVALUATION RESULTS FOR DIFFERENT ENVIRONMENTAL CRITERIA[1)]

		Occurrence & stability of tree layer	Canopy structure and height	Species diversity of tree layer	Forest biodiversity, flora, fauna	Protection function erosion	Hydrology stability and maintenance	soil properties and nutrient recycling	Effect on climate	Use of chemicals; pesticides	Overall rank: total
G = Grasslands allocated to conversion, production and limited production forest											
G1	Sec. forest, TPTI	4	5	5	5	5	6	6	6	6	48
G2	Paraserianthes pl.	2	2	3	3	1	3	3	5	3	25
G3	Pinus plantations	4	2	2	1	1	3	3	2	3	21
G4	Mixed plantations	2	4	4	4	4	5	5	5	4	37
G5	Shorea plantations	5	3	2	3	2	4	4	4	5	32
G6	Teak plantations	5	3	2	2	1	3	3	3	5	27
G7	Permanent grass	1	1	1	1	6	1	1	2	6	20
P = Lowland evergreen rainforest allocated to production forests											
P1	Natural reg. TPTI	4	5	5	5	5	5	5	5	6	45
P2	Paraserianthes pl.	2	2	3	4	3	1	1	3	3	22
P3	Pinus plantations	4	2	2	2	2	1	1	3	3	20
P4	Mixed plantations	2	4	4	4	4	4	2	4	4	32
P5	Shorea plantations	5	3	2	4	4	2	2	2	5	29
P6	Teak plantations	5	3	2	3	2	1	1	3	5	25
P7	Unlogged forest	6	6	6	6	6	6	6	6	6	54
L = Lowland evergreen rainforest allocated to limited production forest											
L1	Natural reg. TPTI	4	5	5	4	5	5	5	5	6	44
L2	Unlogged forest	6	6	6	6	6	6	6	6	6	54

Note: 1) Comparison of the results can be done only within the three area/vegetation categories (G, P & L), and not between them.

TABLE 8.2
POTENTIAL AREAS AVAILABLE FOR DIFFERENT MANAGEMENT ALTERNATIVES
(All figures in 1,000 ha)

Management strategies		Total available	Maximum (40 %)	Minimum (10 %)
G1	Sec. forest, TPTI	7,900	3,160	790
G2	Paraserianthes pl.	7,900	3,160	790
G3	Pinus plantations	6,816	2,726	682
G4	Mixed plantations	6,816	2,726	682
G5	Shorea plantations	6,816	2,726	682
G6	Teak plantations	3,808	1,363	341
G7	Permanent grass	6,816	2,726	682
P1	Natural reg. TPTI	15,453	6,181	1,545
P2	Paraserianthes pl.	5,409	2,163	541
P3	Pinus plantations	5,409	2,163	541
P4	Mixed plantations	5,409	2,163	541
P5	Shorea plantations	5,409	2,163	541
P6	Teak plantations	2,704	1,081	270
P7	Unlogged forests	5,409	2,163	541

Source: Sørensen (1994)

TABLE 8.3
CARBON POTENTIAL ACCUMULATION, ECONOMIC PREFERABILITY AND ENVIRONMENTAL SOUNDNESS OF THE VARIOUS FOREST MANAGEMENT SYSTEMS

Rotation length and management systems	Accumulation potential in 10^6 tons CO_2		Economic preferability		Environmental soundness
	Max. 40%	Min. 10 % (1	Cost efficiency (2	IRR (% p.a.)	
10 years					
G2 (Paras.)	986	246 (25)		29	low
G4 (Mixed)	1,235	309 (31)		14	medium
P4 (Mixed)	837	209 (21)		16	low
30 years					
G3 (Pinus)	1,292	323 (11)	434	10	low
60 years					
G1 (Sec.for.)	711	178 (3)	58	3	high
G5 (Shorea)	3,751	938 (16)		20	medium
G6 (Teak)	1,833	469 (8)		17	medium
P5 (Shorea)	2,228	557 (9)		22	low
P6 (Teak)	1,357	339 (6)		19	low

Notes: (1 Estimates of the mean annual net sequestration of CO_2 during the rotation period are given in parentheses (cf. text).

(2 Measured as kg CO_2 per US$ at 13% p.a. rate of interest, as defined in Eq. (5).

Columns 2 and 3 in Table 8.3 gives the total net accumulation of CO_2 in the standing stock at the end of the rotation period (i.e. before felling) assuming that respectively 10 per cent and 40 per cent of the land area available for the forest management strategies were utilized at the same time for carbon sequestration. If the figure of this net accumulation is divided by the rotation length, we get an estimate of the annual average net carbon sequestration we can expect during the first rotation period, and this estimate is given in parentheses in column 3 of Table 8.3 for the alternative that 10 per cent of the land is utilized. After the first rotation, the net carbon sequestration effect of forestry on that area will be close to zero if the same forest management strategy is maintained.

4. DISCUSSION

4.1 Actions in a 10-year perspective

Table 8.3 shows that strategy G4 (mixed plantation with *Paraserianthes falcataria*, *Acacia magnus* and *Eucalyptus spp.* on grassland) has the largest sequestration potential (because of higher wood densities of the last two species), but the internal rate of return is not as high as for strategy G2 (monoculture with *Paraserianthes falcataria* on grassland). The area suitable for these strategies overlaps considerably. This means that both strategies cannot be implemented at full scale. The monoculture in strategy G2 implies some negative environmental impacts such as low biodiversity, possible degradation of soil and nutrient content, and erosion. With a mixture of other tree species, these problems are reduced in strategy G4, but the economic gain of timber production is much lower in G4 than in G2. If accumulation potential and environmental soundness are given much higher weight than economic profitability, strategy G4 will be preferred as the best action in a 10-year perspective; otherwise strategy G2 is the obvious choice. The annual net carbon fixation of strategy G2 and G4 is respectively 25 and 31 million tons CO_2 for 10 years, assuming that 10 per cent of the available land is used (cf. column 3 of Table 8.3).

Strategy P4 has a low score on environmental soundness and will have severe negative environmental impacts if implemented on virgin lowland evergreen rainforest. Therefore, it is of interest only for logged-over or secondary forests. The strategy has high profitability (16 per cent p.a. internal rate of return) as well as high carbon sequestration potential (about 25 million ton CO_2 per year for 10 years when 10 per cent of the potential area is used), and strategy P4 should be of considerable interest for logged-over or secondary forests.

G4 and P4 together will give a net annual sequestration in the order of 52 million tons CO_2 for 10 years, assuming 10 per cent of the land area available for the strategies is used. If G2 is used instead of G4, the estimate increases to 57 million tons CO_2.

4.2 Action in a perspective of 30 years

Strategy G3, monoculture plantation with *Pinus merkusii*, has high, but not unacceptable, negative environmental impacts. The environmental consequences will be mostly the same as for monoplantation with *Paraserianthes*. The sequestration potential is moderate—only 11 million ton CO_2 per year for 30 years (cf. Table 8.3), which is considerably lower than what strategy G5 has for 60 years. In addition, the profitability of G5 is much higher than for G3, and the former also has better environmental soundness. G3 is therefore not an alternative to recommend.

4.3 Actions in a 60-year perspective

Both *Shorea spp.* and *Tectona grandis* plantations give high sequestration potential. The rates of return are good, and the environmental impacts are moderate. The strategies P5 and P6 will have severe negative environmental impacts if they are implemented on virgin lowland evergreen rainforest, and should only be implemented on logged over or secondary forest. Grassland strategies and production forest strategies are located to different areas, and can possibly be implemented at the same time. A combination of strategy G5 and P5 would give a considerable high sequestration and good economic profit. If 10 per cent of the available land for each of these species were planted, the annual net CO_2 sequestration would be in the order of 25 million tons CO_2 for 60 years.

Strategy G1 has high costs and the carbon sequestration is low. It scores, however, high on environmental soundness, and could therefore, in some areas be of interest.

4.4 Conclusions

The analyses are burdened with considerable uncertainty, both regarding biomass growth, decay time of wood and forest industry products, area potential, and economic data. The site classes chosen are assumed to reflect average situations, which in reality are hard to estimate. It is also assumed an average natural loss as included in the yield tables; in practice such losses vary a lot. The economic data is based on the present economic situation in Indonesia, with relatively low labour costs and high sawwood prices. Also, the analyses are marginal in the sense that no overall administrative costs on regional level are included. If a massive planting program is established, both wood prices and administrative costs may change considerably. As such the analyses in this chapter should be viewed as indicative more than as final results.

Table 8.3 shows that several plantation possibilities exist on grassland which are highly profitable. Most profitable is *Paraserianthes falcataria*, followed by *Shorea spp.* and teak. All of these management strategies are 'no regret' options as they sequester atmospheric carbon without *net* costs. As such they should be implemented in a carbon sequestration strategy. When limited funds are available (which most likely is the case), one could prioritize according to profitability, carbon sequestration profile, and considerations of other ecological factors than carbon sequestration.

It is also seen from Table 8.3 (third column) that the annual net sequestration is highest for the 10-year rotation forest management strategies (21-31 million tons CO_2 per year assuming 10 per cent of the available area is utilized), whereas the corresponding figures for 30-years and 60 years rotations are 11 and 6-16 million tons CO_2, respectively. This illustrates the trade-off between high annual sequestration for a short period versus lower sequestration for a longer period.

Table 8.3 also shows that all the management strategies analysed in this chapter (except strategy G1) are more rewarding than most of profitable methods that are possible to undertake in the Norwegian forestry for sequestration of atmospheric carbon as shown in Solberg and Hoen (1996). This indicates the potential higher gain by investing in forestry in tropical countries than in boreal countries like Norway to reduce the concentration of climate gases in the atmosphere.

It should be emphasized that the thinning strategies assumed and the assumptions that nothing of the wood material is used for the substitution of fossil fuel, underestimate the carbon sequestration estimates in this article. Another factor, which draws in the other direction, is that grassland and soil may accumulate more carbon over time than assumed in this study, and also bring some benefits for local populations. In later analyses, and in possible project implementations, this should be looked into more in detail.

Even though the analyses are burdened with considerable uncertainty, the results of this study illustrate that Indonesia in forestry has a lot of interesting carbon sequestration options which are environmentally acceptable and give high economic return in addition to their carbon off-setting benefits. One main challenge is to get institutional arrangements at the international level so that these options are optimally implemented, to the benefit of all main interest groups involved.

NOTES

[1] This classification is based on the different land categories administered by the Indonesian Ministry of Forestry where harvesting may occur:

i) conversion land (i.e. land which can be designated for agriculture purposes)

ii) production forests

iii) limited production forest.

A more detailed description is given in Lillethun (1994a).

[2] This is rainforest where harvest is permitted. The general conditions are less land sloping, soils not as erodible, and rainfall not so intense as in limited production forests.

[3] This is a way of giving different weight (priority) to CO_2 fixation coming at different times, in addition to the weight implied by r. In this analysis we have set p=0 - i.e. we assume that a marginal change of the concentration of CO_2 has constant value over time independent of what time t it occurs.

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CHAPTER 9

INTERNATIONAL POLICY ISSUES ON CARBON FLUXES AND FORESTS IN THE SOUTH

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1. ABATEMENT OF CO_2 EMISSIONS AND THE DEVELOPING WORLD

The differences in historical cumulative share of carbon dioxide (CO_2) emissions among countries reflect their unequal patterns of economic development. Developed countries benefit today from the accumulation of man-made assets made mainly possible by the industrial revolution and the subsequent pattern of high levels of fossil fuels consumption and CO_2 emissions. In contrast compared to the developed countries, the economic performance of developing countries was relatively poor, as was their energy consumption. According to estimates, developing countries were responsible for only 15 per cent of the total CO_2 emissions between 1870 and 1986 (Sathaye and Reddy 1993).

Unfortunately, technology has not yet advanced to the stage where economic development could be achieved without fossil fuels consumption. As a former secretary-general of UNCTAD acknowledges:

> Although there is no reason to believe that completion of this process (of industrialization) requires other countries to follow step by step the paths established by developed countries, the broad characteristics of the process of development are clear enough: development means industrialization; and industrialization entails the systematic application of controlled mechanical energy to an ever-widening range of productive activities (UNCTAD 1992: ii).

In other words, there is an apparent conflict between the objectives of reducing CO_2 emissions and closing the socioeconomic gap between developed and developing countries. Developing countries need to increase their average energy consumption at faster rates than the developed world if the gap between their respective production and consumption patterns is to be reduced. However, the costs of abatement of greenhouse gases (GHG) emissions are considerably smaller in the developing countries. Moreover, their share in total emissions is increasing with time, and some developing countries are among those most affected by possible damages introduced by the climate change—thus, developing countries are an essential component of any effective international strategy aiming at emissions abatement.

The biggest challenge presented by the GHG abatement agreements is to conciliate *equity* and *efficiency* in a common strategy. The principle that has been used so far in similar agreements (for example, the Montreal Protocol), based on 'grandfathering' rights (i.e., future levels of emission should be determined as a function of the current level), cannot be accepted under these terms. Equal percentage reductions are intrinsically unfair since they freeze the *status quo* of energy and resource consumption (which is widely recognized as an unfair distribution between rich and poor countries). This would also penalize the economies that are presently investing to reduce the emission intensity of their production (emissions per unit of output)—these economies could be 'giving away' future emission rights through current abatement measures. Finally, emissions abatement based on uniform cuts does not respect the efficiency principle since the associated marginal costs of emission reductions vary considerably among countries.

There are also 'altruistic' proposals which emphasize that only developed countries should be penalized by their past and present higher levels of emission. However, these proposals would face two main problems. First, there would be great resistance from developed countries to accept the idea that massive transfer payments, without reciprocal efforts, could be sent to developing countries, which makes the proposal unacceptable in political terms. Second, the impacts of the reduction in economic activities in the developed world would have considerable social costs, affecting the welfare of the entire human population. For instance, a reduction in the production of goods would affect exports to the third world, making them more scarce and expensive, while at the same time, imports from the developing countries would be considerably reduced. The experience of the 1970s illustrates this problem: the economic retraction of the developed countries induced a world recession with dramatic consequences to developing countries (reduction of primary goods exports combined with the external debt crisis), some of which are still paying the costs of adjusting to the new world order.

Therefore, the establishment of international instruments to limit CO_2 emissions should be considered in a context of open economies—that is, recognizing that international flows of income have background effects to national economies. This can be exemplified in a very simple situation. Consider two countries (A, B) with different costs of abatement, but the one with higher costs of abatement (A) being politically interested in the control of greenhouse gases. It is possible that a deal between both countries is reached: country A is willing to pay compensation to reduce emissions in country B up to a point where the marginal contribution equals the benefit of reducing GHG emissions (for country A); similarly, country B is willing to accept compensation to reduce its economic activity up to a point where the marginal compensation equals the marginal cost of sacrificed production. The use of economic instruments allows these transfers to take place in a flexible, cost-effective way. The options currently discussed to implement them are examined in the next section.

2. REDUCING CO_2 EMISSIONS: THE EXISTING PROPOSALS

The Framework Convention on Climate Change was signed in June 1992 in Rio de Janeiro. Since the divergence among the participant countries at the time was considerable, the terms of the Convention were left mostly unspecified in order to allow negotiations to take over the coming years. The most important target, the 'stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system' (Article 2), has been disconnected from detailed targets and timetables. Only developed countries and a small number of emerging developing countries have committed themselves to the 'return by the end of the decade to earlier levels of anthropogenic emissions of carbon dioxide and other greenhouse gases' (Article 4), which is commonly associated with stabilizing CO_2 emissions by the year 2000 at the 1990 level. However, national targets for CO_2 abatement adopted voluntarily by some European countries are more strict than these (for example, Denmark has an ambitious target level of 20 per cent CO_2 reduction of the 1988 level by the year 2005; see Svendsen 1996).

The Convention also establishes the Global Environmental Facility (GEF) as the financial mechanism for the provision of resources on a grant or concessional basis. The programme priorities and eligibility criteria were left to be determined by the Conference of the Parties, and there was also a vague reference to financial resources related to the implementation of bilateral, regional, and multilateral channels between developed and developing countries.

The flexibility of the terms used in the Convention reflects the uncertainty about the real commitment and political willingness of each participant country to deal with the adjustments imposed by the need for emissions reduction. This also means that there is a wide range of implementation options if developed countries effectively decide to pay financial compensations to developing countries in order to 'import' CO_2 abatement from developing countries.

The literature has emphasized economic instruments: carbon taxes, external offsets and tradable permits. These mechanisms respect the 'polluter-pays' principle, internalizing pollution costs, but providing flexibility to the economic agents to minimize costs in the adjustment to lower levels of emissions, so that:

> they separate the question of where abatement can be undertaken (hopefully, at the lowest cost), from the question of who should pay, which is determined by the initial allocation/targets, or tax redistributions' (Grubb 1992: 11).

Nevertheless, there are important differences in their implementation. A system of international emission taxes can be established with revenues from carbon taxes being collected in countries with high levels of emission, and transferred to countries which preserve carbon sinks. The essence of the tax is to encourage tax-avoiding behaviour, through the permanent incentive of energy conservation, substitution of non-carbon energy for carbon energy, and any other innovation that reduces pollution (Pearce 1992).

Special allowances can be made to prevent poor countries from becoming net payers of carbon taxes, respecting the equity principle.

Current experiences show that the establishment of international carbon taxes is not problem free. One point in particular deserves special attention: there is considerable uncertainty about the effective reduction in emissions related to a certain tax level. Therefore policy makers have to guess the tax level necessary to achieve the desired target in emissions reduction. Moreover, if this tax level is high, there will be strong resistance from penalized economic agents to accept the increase in their costs (creating the risks of unemployment and declining competitiveness). This problem is highlighted by Svendsen (1996) in the analysis of the Danish experience: the CO_2 tax implemented in 1992 (households pay US$ 16 per ton of CO_2 emitted, and VAT-registered firms pay half the tax) is considered too low to achieve the politically-decided target of 20 per cent reduction by the year 2005—it is estimated that this target can only be achieved if the tax is raised to US$ 50.

Finally, at the international level, penalizing countries that do not respect the agreements (for example, not paying the stipulated tax amounts) is a formidable problem. An international agency responsible for monitoring and enforcement would not have the necessary power to obligate all countries to adhere to its decisions and, as agreements are voluntary, too much pressure could induce the withdrawal of the transgressor country (a worse-off solution).

In any case, even if emission targets are not achieved, carbon taxes are a very efficient system to obtain revenues earmarked for environmental action. For instance, Svendsen (1996) argues that the main motivation behind the Danish initiative of taxing CO_2 emissions is to 'justify'—through 'green' objectives—higher levels of taxation relative to the European Union standards. Therefore, conventional taxes could be lowered without affecting the total tax revenue (that is, a double-dividend strategy).

This property is very important if carbon taxes are to be coupled with funding abatement schemes in other countries. Indeed, this is the main idea behind external offsets: a country can meet its own national emission target by reducing emissions domestically or 'offset' this by investments to reduce emissions in equivalent amounts elsewhere, so that the total of national emissions minus the external 'savings' would meet the target (Grubb 1992).

The implicit assumption is that developing countries have a large stock of economically inefficient CO_2 sources (that is, economic returns per unit of CO_2 are lower than in developed countries) with cheap abatement options. This would bring flexibility to developed countries in the process of reducing their own CO_2 emissions, while at the same time create an economic value for the forestry carbon fixation capacity preserved in developing countries. In other words, the CO_2 abatement services currently provided freely from the south to the north would be considered as a kind of export service, thus to be paid for.

The establishment of an international system of CO_2 emissions reduction offset would require that countries commit themselves to emission reduction targets which are mutually respected, accepting the principle that emission savings can be obtained through abatement investment in other countries. The responsibility to meet the abatement commitments can be transferred onto public and private entities within these countries. This means that abatement services can also be provided by private entities, not necessarily requiring interstate agreement. In fact, there are already examples of jointly implemented forestry offset projects funded by private US electricity utilities which invested in carbon fixing reforestation in developing countries (Hayes 1993; IPCC 1996).

From the analysis above, it is clear that external offsets have great potential as a transfer mechanism to compensate the developing countries which decide against depleting their natural forests (or, at least, to reduce the deforestation process). In the long term, however, one problem remains. The system is based on transfers to developing countries independently of the efforts being made to reduce emissions outside the abatement projects. For example, one country may receive compensation because it has preserved natural forests, but at the same time, other emission sources have been accelerated (in the energy, industry or transportation sectors). Since these other sources of emissions are expected to increase in developing countries, the external offsets cannot be considered a long-term global solution. Furthermore, it is expected that the volume of resources to be transferred from developed countries to developing countries could be substantial, arousing strong opposition from net payers in the north (the situation in some industrialized countries could be worse off after the transfers than before; see Pearce 1992). Therefore, many authors consider that external offsets can only be the first stage of the implementation of international tradable emission permits.

Tradable permits have the advantage of directly establishing a maximum amount of emissions for the system as a whole (taxes, for example, can only affect the total level of emission indirectly). At the same time, relevant agents have the freedom to adopt their own cost structures, thus assuring economic efficiency.

The experience of the United States is an obligatory reference with regard to the feasibility of tradable permits (for a review, see Svendsen 1996). Some of the permit programmes were established at the federal level by the US Environmental Protection Agency (EPA), all of them dealing with air pollution. The first programme was the Emission Trading Programme (started in 1974), aiming for the industry to achieve national air quality standards on sulphur oxides, nitrogen oxides, particulates, carbon monoxide, hydrocarbons, lead and ozone. In the 1980s, two similar programmes were established to phase-out leaded gasoline (1982-87); and chlorofluorocarbons (CFC) and halons (1989-96). The most recent experience is already considered a successful example of cost-effective pollution control strategy: the Acid Rain Programme, aiming at the reduction of sulphur dioxide emissions by electrical utilities (50 per cent reduction of the 1980 level by the year 2000). Other programmes, managed by state agencies, deal with air and water pollution control. An important feature of all US programmes is the initial concession of permits based on past emission levels (grandfathering).

The advantages of tradable permits include the flexibility to reduce emissions at low costs, and the permanent incentive for users to invest in technological innovation that reduces the need to pay for additional permits. At the international level, a permit market can be considered as a system according to which one country reduces its emissions in favour of compensation paid by another that continues to emit, provided that the sum of emissions of the two countries do not exceed their combined quotas. Such a system would allow net transfers to developing countries if credits are provided for the preservation of native or new forests as carbon sinks.

However, the implementation of tradable permits is the subject of much criticism. First, as in the case of carbon taxes, the need of an agency to oversee the trade in permits does not solve the problem of penalty enforcement for countries exceeding their emission quotas. Second, the international permit market would be subject to many imperfections, since there is a great imbalance between the size of the potential buyers (large economies) and sellers (mostly small economies). Third, and probably the most crucial, is the virtual impossibility of reaching a consensus over the definition of initial quotas. There is considerable debate about the best way of distributing the permits. On the one hand, the proposal of 'grandfathering' the permits according to the existing levels of emissions (as in the US experience) would not be acceptable to the developing countries, since this approach does not respect the equity principle (in the sense that it benefits the currently high-pollutant countries). On the other hand, proposals aiming at 'fair' allocation, such as granting quotas proportionally to the number of inhabitants (the idea of an equal, individual right to emit) would face great resistance from the largest industrialized countries.

It should be noted that the experiences reviewed above refer to domestic programmes, so there is no assurance that some of their successful aspects can also be achieved in the international context. Also, as argued by Pearce (1994), most studies assume implicitly that costs of carbon emission controls vary significantly between trading parties, and that transaction costs do not represent an obstacle. The first point is challenged by the fact that developed countries with better availability of capital and technology, may face equal or even lower abatement costs than poorer countries. The second point refers to the problem that the 'thinner' the market the higher the transaction costs, leading to a paradox: an effective joint implementation programme requires large-scale trading; however, large-scale trading cannot occur until it has been demonstrated the efficiency of joint implementation.

Another institutional problem emphasized by Pearce (1994) refers to the definition of what can be considered 'genuine' compensation for CO_2 emissions reduction. The Climate Change Convention (and the Montreal Protocol) considers that emission control obligations can be jointly implemented by developed countries and other parties. Joint implementation can be obtained through compensations from donors to countries that host carbon abatement programmes, and compensation values should be determined by the 'agreed full incremental costs' of these programmes. However, there is no proper definition of 'full incremental costs'. It is understood that incremental costs should compare actual results against a counterfactual baseline (what they would have done if the emission control programmes did not take place). This is a particularly complicated task, especially because the Convention did not establish reduction targets for developing countries. Pearce (1994) argues that there is no real need for an explicit set of guidelines on how to estimate incremental costs since 'donor' and 'host' will agree a price, each of whom will know whether or not the arrangement is in their advantage. Nevertheless, as discussed before, the 'market' for CO_2 transactions is far from a perfect competition hypothesis. It is possible that a few industrialized countries, acting in cooperation, will have much more power in the negotiations than a large number of indebted and atomized developing countries. A possible result is that opportunity costs would be omitted from these negotiations, thereby reducing compensation values and creating an implicit subsidy to developed countries. The relevance of opportunity costs is discussed in the next sections.

From the discussion above, it is clear that difficulties in the implementation of economic instruments to control CO_2 emissions at the global scale will require a long process of negotiation. An effective system will probably combine some of many different proposals, mixing command-and-control and market-oriented tools. National strategies in developed and developing countries might contemplate carbon emission taxes in order to encourage a permanent stream of innovations that strive for the reduction of marginal costs of abatement for all concerned. Offset investments in abatement might be considered by developed countries to take advantage of lower marginal abatement costs in developing countries. Finally, tradable permits might be used to consolidate the system operation once it has reached a mature stage when the costs and rewards of reducing CO_2 emissions are relatively clear to most nations.

Time will certainly be needed for the elaboration of a joint framework whereby the historical responsibility of developed countries can be matched by their willingness to pay, and the current pattern of deforestation in developing countries can be reversed by the potential benefits of conservation. Nevertheless, this does not mean that initial steps cannot be taken. One of these steps is the establishment of compensation schemes for developing countries that effectively invest in forest preservation, thus averting an increase in CO_2 emissions. The next section deals in more detail with the great potential of abatement schemes based on north-south transfers.

3. TROPICAL FORESTS AND CO_2 EMISSIONS

The future of tropical forests will be decisive in the expansion or reduction of the concentration of CO_2 in the atmosphere. Forests are natural sinks of carbon, and the current trend of forest depletion and degradation in the tropics is a substantial source of CO_2 and other GHG emissions—in these areas, deforestation is usually associated with the burning of biomass, which liberates gases into the atmosphere. The International Panel on Climate Change (IPCC 1996) estimates that low-latitude forests were responsible for an annual increase of approximately 1.6 Gt of carbon to the atmosphere by 1990. Considering that the annual rate of deforestation is about 15.4 million hectares, this corresponds to one hectare of deforestation in a low-latitude country producing, on average, a net flux of around one hundred tons of carbon.

Therefore, slowing down the current rates of tropical deforestation would represent a significant contribution to controlling carbon dioxide and other GHG emissions. In the medium term, if forest management options are economically feasible, the role of the tropics in the balance of GHGs can be reversed through the expansion of forestry activities with net gains in the storage of carbon. In forestry expansion, two elements should be considered: the potential capacity for carbon storage and the costs of mitigating options.

According to IPCC (1996) estimates, the potential of low-latitude forests to conserve and sequester carbon ranges between 45 to 72 Gtons, more than half of which would result from promoting forest regeneration and slowing down deforestation. This is considerably larger than the potential of temperate and boreal zones: about 13 Gtons and 2.4 Gtons, respectively. Tropical America has the largest potential for carbon conservation and sequestration (46 per cent of tropical total), followed by tropical Asia (34 per cent) and tropical Africa (20 per cent).

TABLE 9.1
COSTS OF FOREST PROTECTION OR REDUCTION DEFORESTATION

Country	Cost ($/t C)	Source and notes
Brazil	2.3 [a] - 4 [b]	[a] Darmstadter and Plantinga (1991) [b] Cline (1993)
Côte d'Ivore	8	Darmstadter and Plantinga (1991)
Indonesia	15	Darmstadter and Plantinga (1991)
Thailand	0.4 - 0.8	Based on Wangwacharakul and Bowonwiwat (1995), which includes government budget for protection and opportunity cost of land for agriculture production
Mexico	1 - 6	Based on data in Masera *et al.* (1995); lower bound on cost of protection of tropical evergreen forest of Tabasco
India	0.5	Based on $5/ha cost for a tiger sanctuary and 50t C/ha of biomass density
Central America	1 - 3	Swisher (1991) estimate, based on cost of protected areas reported in the Tropical Forest Action Plan (TFAP) proposals for Cost Rica, Honduras, and Panama
Russia	1 - 3	Krankina and Dixon (1994)

Source: IPCC (1996)

The potential gains in carbon storage are considerable. The IPCC (1996) baseline scenario (under current climate conditions and assuming no change in the estimated available lands over the period of interest) estimates that the cumulative amount of carbon that could potentially be conserved and sequestered over the period 1995-2050 by slowing deforestation (138 million hectares) and promoting natural forest regeneration (217 million hectares) in the tropics, combined with the implementation of a global forestation programme (345 million hectares of plantations and agroforestry), would be equivalent to 12-15 per cent of the cumulative fossil fuel emissions of carbon over the same period, projected according to the IPCC (1992) scenario.

The other relevant dimension refers to the (reportedly) low costs of carbon conservation and sequestration projects in the tropics. These costs usually consider only the direct costs of forest projects (protection, management, etc.) but not the opportunity cost of the land. Therefore, there is a bias to present mitigating strategies in developing countries considerably cheaper than in developed countries. The consequences of this problem are discussed later.

The IPCC review on the costs of forest protection and deforestation reduction followed the pattern of considering only direct costs of carbon sequestration—the opportunity cost of land was omitted (Table 9.1). The results ranged from about US$ 8/ton carbon for afforestation and reduction of deforestation in the tropics, increasing to about US$ 28/ton carbon for afforestation in non-US OECD countries. The costs for establishing a forest plantation (opportunity cost of land excluded) were estimated to range between US$ 230 and US$ 1,000 per hectare, with an average cost of US$ 400 per hectare.

The 1996 IPCC report surveys more recent studies estimating the costs of preserving and expanding carbon sinks, including the benefits that forestry options would present for dwellers. Opportunity costs of land, however, remained omitted. The consequence is that most of the initial costs of expanding carbon sinks are smaller than the ones presented in the previous report (Table 9.2).

TABLE 9.2
INITIAL COST OF EXPANDING CARBON SINKS
BY DIFFERENT REGIONS AND PRACTICES

Region/country	Practice	Cost [a] (US$/t C)	Source
Boreal	Natural regeneration [b]	5 (4-11)	Dixon *et al.* (1994)
	Reforestation	8 (3-27)	
Temperate	Natural regeneration [b]	1	Dixon *et al.* (1994)
	Afforestation	2 (1-5)	
	Reforestation	6 (3-29)	
Tropical	Natural regeneration [b]	1 (1-2)	Dixon *et al.* (1994)
	Agroforestry	5 (b-11)	
	Reforestation	7 (3-29)	
Central America	Regeneration	4	Swisher (1991)
	Agroforestry	4	
	Plantations	13	
Argentina	Reforestation	31	Winjum *et al.* (1993)
	Afforestation	18	
Australia	Reforestation	5	Winjum *et al.* (1993)
Brazil	Reforestation	10	Winjum *et al.* (1993)
	FLORAN	3-8 [c]	Andrasko *et al.* (1991)
Canada	Reforestation	11	Winjum *et al.* (1993)
	Regeneration	6	

Table 9.2 (con't)

Region/country	Practice	Cost [a] (US$/t C)	Source
China	Reforestation	10	Winjum *et al.* (1993)
	Forest management	3-4	Xu (1995)
	Eucalyptus plantations	8	
	Agroforestry	6-21	
Germany	Reforestation	29	Winjum *et al.* (1993)
India	Reforestation	15	Winjum *et al.* (1993)
	Regeneration	2	Ravindranath *et al.* (1995)
	Teak plantation	3	
	Agroforestry	9	
Malaysia	Reforestation	5	Winjum *et al.* (1993)
Mexico	Reforestation	4	Winjum *et al.* (1993)
	Plantations	5-11	Masera *et al.* (1995)
	Forest management	03-3	
South Africa	Reforestation	9	Winjum *et al.* (1993)
Thailand	Teak plantations	13-26	Wangwacharakul and Bowonwiwat (1995)
	Eucalyptus plantations	5-8	
	Agroforestry	8-12	
USA	Reforestation	5	Winjum *et al.* (1993)
	Afforestation	2	
	Various options	5-43 [d]	Moulton and Richards (1990)
	Various options	19-95 [e]	Adams *et al.* (1993)
Former Soviet Union	Reforestation	6	Winjum *et al.* (1993)
	Regeneration	5	
Russia	Plantation	1-8	Krankina and Dixon (1994)

Source: IPCC (1996)

Notes:
[a] Forest components for sequestering carbon vary by source: Dixon *et al.* (1994), Krankina and Dixon (1994), and Winjum *et al.* (1993) include only carbon in vegetation; Xu (1995), Ravindranath and Somashekhar (1995), Wangwacharakul and Bowonwiwat (1995), and Masera *et al.* (1995) include vegetation and soil carbon; Swisher (1991), Moulton and Richards (1990), and Adams *et al.* (1993) account for C in vegetation, soil, and litter.
[b] Values in parentheses are interquartile ranges.
[c] Figures vary depending on land rental costs per ha from US$ 400 to US$ 1,000; Floram=Florestales Amazonia.
[d] Marginal costs include planting and land rental costs.
[e] Includes land rental costs.

The IPCC reports and other studies suggest that land prices should be used as *proxies* for the opportunity cost of land. In many areas across developing countries, this price would be close to zero, as in the case of degraded land suitable for reforestation. Unfortunately, however, land productivity is not the only factor affecting land prices. Uncertainty concerning property rights, abundant supply of 'quasi-open' access land in the frontier between the forest and agricultural areas, lack of capital to invest in proper management, non-existent credit schemes for small farmers and many other factors negatively affect land prices (for an overall analysis of the causes of deforestation in developing countries, see Palo and Mery 1996). The consequence is that land prices do not accurately represent the opportunity cost of land: if a system of international transfer payments for arresting agricultural development in tropical forests is based on existing land prices, there would be an implicit subsidy from developing countries to developed countries because land prices underestimate the opportunity cost of land. The discussion of this problem in the context of the Amazon is analysed in the next section.

4. THE OPPORTUNITY COST OF FOREST CONSERVATION: EVIDENCES FROM THE AMAZON

According to the theory of economics, if there are no market distortions, actual prices are the proper measure of the opportunity cost of an asset. Based on this principle, many studies consider land prices as a proxy for the opportunity cost of land (as in the case of the IPCC report, discussed above).

In a similar way, a World Bank dissemination paper (Schneider 1993) argues that there could be considerable gains from the trade in carbon emissions between industrialized countries and landowners and governments in the Amazon. The methodology was based on a comparison of the global benefit per hectare of forest as a store of carbon (combining the amount of carbon sequestered in a hectare of forest and the per-ton value to society of reducing carbon emission) and its value as agricultural land, assuming that:

> The value of forest land in agricultural use is best estimated by the selling price of forest land' (Schneider 1993: 1)

This approach assumes implicitly that any positive difference between the benefits of carbon sequestering (which allows extra economic activity in the north) and the compensation payments based on actual forest prices in the south would accrue as consumer surplus to the payers (developed countries). Indeed, it is the disparity between these values that Schneider (1993) considers an encouragement to the proposal of compensation schemes. In his estimates, the value of one hectare of Amazonian forest land range from US$ 600 to US$ 7,000 (based on the value that industrialized societies have actually demonstrated as their willingness to pay for carbon sequestration) while actual land prices vary from US$ 2.5/ha to US$ 300/ha.

However, there are considerable distortions in land markets in the Amazon. Among other factors, the abundance of quasi-open access land in the frontier, coupled with the insecurity over property rights of land already occupied, produces a negative bias to land prices as an indicator of the opportunity cost of land. Hence, payments to arrest the existing patterns of economic development in the Amazon based on current land prices represent an implicit subsidy to the developed world, since developing countries would not get decent 'value for their money'. In other words, Amazonian countries could be selling the services of CO_2 sequestration at a price smaller than the flow of future benefits they could expect from the land, had the forest not been preserved. This would not respect the equity principle, since most of the developed world is already receiving economic benefits from higher patterns of CO_2 emissions.

In order to illustrate the problem of undervaluation, a few valuation studies in the Amazon are reviewed in this section. The selected studies estimate the economic returns of alternative land-use options that can be considered as better proxies of opportunity costs of land than land prices. These include Peters *et al*. (1989) whose study concerns the valuation of one hectare of forest reserve in Mishana, Peruvian Amazon; Pinedo-Vasquez *et al.* (1992) who analyse the local population's land-use options in one hectare of forest reserve in San Rafael, Peruvian Amazon; Toniolo and Uhl (1992) who deal with commercial agriculture options in Uraim, a state of Pará (southeastern Brazilian Amazon) and Uhl *et al.* (1991) on land-use options in Tailândia, a state of Pará.

TABLE 9.3
NET PRESENT VALUE ESTIMATES OF ONE HECTARE OF LAND
(discount rate: 5 per cent)

Peru (Mishana)	selective logging plus extractivism	6820,00
Eastern Amazon [a]	commercial agriculture plus dairy herd	3974,07
Peru (San Rafael)	one-time timber removal plus swidden agriculture	2997,88
Eastern Amazon [b]	one-time timber removal plus dairy production (intensive)	1385,19
Peru (San Rafael)	one-time timber removal plus extractivism	741,90
Eastern Amazon [b]	one-time timber removal plus dairy production (extensive)	624,92
Eastern Amazon [c]	one-time timber removal plus beef production (extensive)	282,20
Eastern Amazon (Pará)	average forested land price (1989) [d]	252,30
Eastern Amazon [e]	selective logging	214,07

Notes: [a] Based on the average composition of a farm in Uraim (Toniolo and Uhl 1992).
[b] Combining data for one-time timber removal in the Paragominas region (Veríssimo *et al.* 1992) and dairy herd productivity in Uraim (Toniolo and Uhl 1992).
[c] Combining data for one-time timber removal in the Paragominas region (Veríssimo *et al.* 1992) and extensive ranching in Paragominas (Mattos and Uhl 1994).
[d] Based on land prices data collected by Fundação Getúlio Vargas.
[e] Based on data from Tailândia (Uhl *et al.* 1991).

Valuation studies are carried out according to different assumptions and methodologies. Moreover, each site presents many ecological and socioeconomic specificities (for example, soil quality, topography, biodiversity, economic infrastructure, location, migration flows, etc.), making the comparison of results not an easy task. The choice for these studies was based on the possibility of comparing relatively closely-located sites (Mishana and San Rafael; Uraim and Tailândia) and data availability that allows the adoption of a standard methodological approach.

The exercise was based on the estimate of net present values of alternative land-use options in one hectare of forested land, assuming a 5 per cent discount rate. Prices are expressed in 1989 US dollars. Table 9.3 above presents the results for the selected studies.

It is clear that the returns from sustainable extraction of timber and non-timber products obtained by Peters *et al.* (1989) in Mishana (Peruvian Amazon) are exceptionally high. This study is the most referred to valuation exercise for the Amazon forest, probably because the high returns from forestry and extractivist activities appear to justify the widespread optimism among environmentalists that greater profit can be achieved with sustainable activities which retain the forests than with the conventional agricultural practices. The high returns are explained by the unusual composition of tree species: the Mishana forest reserve presents a concentration of high value species that is not representative of other areas in the Amazon. Indeed, the data would suggest a process of forest 'enrichment', with very high incidence of commercial fruits and high value timber species. For example, in only one hectare, there are 83 trees of the *Iryanthera, Virola* family, a high commercial value timber, and 36 trees of the *Jesenia batava (mart.) Burret* species, a fruit tree with high productivity. Another unusual characteristic of the site is its proximity to Iquitos, a major urban centre, thus considerably reducing transportation costs.

The net present value expected (NPV) from fruit extraction was estimated at US$ 6,300, and US$ 490 for the NPV from selective logging, for a total NPV of US$ 6,820 for economic activities not requiring forest depletion. According to this result, since local net benefits from the conservation of the forest are higher than from any activity associated with deforestation, there is justification for transfer payments based on carbon conservation and sequestering. In other words, the current deforestation process cannot be explained by rational economic behaviour, as net returns from preserving the forest are higher than those obtained by its depletion.

Pinedo-Vasquez *et al.* (1992) used the same approach—inventory of tree species in one hectare of forest, but in a site resembling more closely the average situation of Amazon forests. San Rafael is not located as advantageous to an urban centre (Iquitos) as Mishana, and there is no spectacular concentration of species with high commercial value. The results indicated a conflicting conclusion: current agricultural practices (NPV=US$ 2,516.94) coupled with one-time harvesting of all merchantable timber (NPV=US$ 480.94) are considerably more attractive to local dwellers than collection of fruits and latexes (NPV=US$ 399.40) or selective timber (NPV=US$ 342.50). Two important findings were the observations that only half of the dozen commercial fruit and latex species inventoried in Mishana were available in San Rafael, and there was no harvesting of fruit or latex from the reserve for the Iquitos market, even though additional species with potential markets similar to those identified in the Mishana reserve were present. These results would justify the behaviour of local populations in economic terms, given their capital endowment and time-horizon:

> Examining the actual land-use choices made by the San Rafael population emphasizes the logic of their decision making within the current context in which rural population prioritize their economic activities. Within that context, *ribereños* can be expected to continue converting forested land to swidden agriculture unless alternative land uses become more attractive economically (Pinedo-Vasquez *et al.* 1992: 172).

A better comparison of alternative land-use options is provided by a number of valuation exercises in the Paragominas region, state of Pará (Brazilian Eastern Amazon). The studies covered different production conditions, from capital-intensive agriculture and dairy production activities to extensive ranching and timber extraction, reproducing the economic rationality of distinct land-use decisions.

The highest productivity was obtained in Uraim. The site is located in a region where the first settlers established themselves as small- and medium-sized farmers in the early 1960s, and is close to the town of Paragominas and the Belém-Brasília road. These special features resulted in a process of agricultural intensification and diversification that is not typical in other Amazon areas (Toniolo and Uhl 1992). Therefore, the returns from agriculture are considerably higher than the Amazonian average. The distribution of productive land in Uraim is as follows: 59.77 per cent for cattle (dairy herd), 12.77 per cent for shifting cultivation, 0.25 per cent for vegetable cropping, and 27.21 per cent for perennial crops. The relevance of perennials (black peppers and oranges), which require a time lag between investment and harvest, leads to the importance of the discount rate option for ultimate results. Both perennial crops require high investments at the beginning; moreover, the first harvest is available only after two or three year. Higher discount rates mean that the net present value of future benefits is reduced (or, symmetrically, higher capital costs per unit of output), thus reducing the profitability of cultivation.

This point illustrates the relevance of capital availability in valuation exercises. Usually, all measures are calculated as per unit of area (hectare). Nevertheless, this equals to considering that land is a scarce resource—indeed, in order to properly assess differences among land-use options which are more or less intensive in each production factor, proper valuation exercises should also consider measures per unit of labour and per unit of capital invested. The high returns obtained for Uraim are the consequence of previous investments that necessitated some sort of capital endowment or access to credit facilities. These usually do not exist for most small farmers in the Amazon. Therefore, the opportunity cost of land is a function of credit access, and the true potential of future revenues should be considered in a context that has solved this market failure.

Combining data obtained from Uraim (which include land clearance and other investment requirements) to the net returns from one-time timber harvesting in the Paragominas county (Veríssimo *et al.* 1991), it was possible to estimate the net returns from deforestation followed by intensive or semi-intensive dairy ranching. A similar exercise was carried out for extensive beef production (using data from Mattos and Uhl 1994). All obtained results were above the average price for forested land in the state of Pará, showing that actual land prices underestimate the economic potential of the land. This problem is more important in capital-intensive activities, and suggests that the opportunity cost of reducing agricultural expansion is considerable for the local economy.

The only activity that remained below the average land price was selective logging. The data, extracted from sites near the town of Tailândia and a paved highway (Uhl *et al.* 1991), are based on a 20-year cycle. No data were available for non-timber forest products, therefore it was not possible to estimate the proper NPV of extractivism. Nevertheless, the absence of information suggests that non-timber forest products are not financially relevant to local farmers. Therefore, compensations to locals are justified if agricultural expansion is reduced in the region for the purpose of abating the global CO_2 emission problem.

5. THE OPPORTUNITY COST OF CURBING AGRICULTURAL ACTIVITIES IN THE BRAZILIAN AMAZON: A SIMULATION EXERCISE

The previous section shows that reducing commercial agricultural expansion in the Amazon would result in a positive opportunity cost for the local economy. However, it is somewhat questionable to use values obtained from a limited number of site studies for evaluating the entire Amazon region because the particularities of the sites examined may introduce major biases into the analysis.

In order to provide some figures for considering the Brazilian Amazon as an entity, some projections were obtained by a simulation model based on the DESMAT database. The simulation model was originally elaborated by Reis and Margulis (1991) to estimate global warming impacts of deforestation in the Amazon, and then reviewed to estimate the deforestation impacts of the Carajás iron project in the Northeastern Brazilian Amazon (Reis 1996). DESMAT consolidates economic, social and ecological data at municipal and state levels for the Brazilian Amazon, from censuses and other surveys being applied in many empirical exercises.

TABLE 9.4
AGGREGATE RESULTS OF THE SIMULATION MODEL (REFERENCE SCENARIO)

	1990	1995	2000	2005	2010
Agriculture GDP (US$ millions)	2,832	3,469	4,116	4,689	5,149
Total GDP (US$ millions)	20,549	24,467	29,141	34,898	41,627
Accumulated carbon emissions (tons, millions)	5,697	7,033	8,226	9,255	10,090
Agricultural land (hectares, millions)	56.6	68.4	79.3	88.7	96.6

Source: Reis (1996)

The simulation model combines projected changes in road structure to demographic, economic and ecological variables that are time-lagged.[1] Total increase in the road network is determined exogenously according to plans for road expansion in each municipality. However, the distribution between paved and non-paved roads is determined endogenously by a function considering (lagged) economic, demographic and ecological variables. Similar procedures are used to estimate demographic and economic variables. For example, population growth is a function of the previous trend in population growth itself, plus economic growth and other socioeconomic variables. Deforestation is defined by the net changes in agricultural areas (including fallow lands).

Carbon emissions are estimated according to expected changes in land use (including average fallow time), biomass and carbon content parameters. The most important results are shown in Table 9.4.

It is assumed that the opportunity cost of land is determined by the potential agricultural product to be sacrificed if forest areas were reserved for carbon conservation. Therefore, (average) opportunity costs can be estimated by dividing the changes in carbon emissions and the changes in total GDP for each period. Two estimates are provided: agricultural GDP and total GDP. Total GDP is considered in the analysis because agriculture has important multiplier effects in urban activities—a large part of urban GDP probably could not be generated without accompanying land clearance (Andersen 1996). The 'true' opportunity cost for the entire Amazon economy lies between these two values. The results are shown in Table 9.5.

TABLE 9.5
OPPORTUNITY COSTS AND CARBON EMISSION CONTENTS

	1990	1995	2000	2005	2010
Agricultural GDP: US$/ton C	7.1	12.0	16.2	21.7	29.8
Agricultural GDP: US$/ha	959	1,359	1,772	2,359	3,160
Total GDP: US$/ton C	51.8	84.3	114.8	161.9	240.9
Total GDP: US$/ha	6,958	9,586	12,548	17,555	25,548
Emission per area (ton C/ha)	134	114	109	108	106

It is clear that the strategy of conserving carbon in forestlands would result in increased opportunity costs over time: the value for the year 2010 (US$ 29.8/ton for the agricultural GDP only, and US$ 240.9 for the total GDP) is more than four times the initial value of US$ 2.79/ton. These costs increase with GDP, indicating that the opportunity cost is a function of the level of economic growth. Net emissions per hectare are close to Schneider's estimates (ranging from 134 tons/ha to 106 tons/ha), indicating a declining trend. The combination of both results provide the opportunity costs per hectare: the opportunity costs start at US$ 959 for 1990 when only the loss of agricultural output is considered and reach US$ 3,160 by 2010—even the lowest value exceeds, by far, the best land price considered by Schneider (1993).

Note that the annual rate of growth of the opportunity cost of forest land, measured in terms of expected agricultural output growth, is 6.1 per cent per annum (6.7 per cent if total GDP is considered). This value exceeds the usual range assumed for the social discount rate (2 per cent to 5 per cent), showing that, even in present value terms, the opportunity cost of the foregone output would still increase in time (Table 9.6).

In other words, even if discounting is considered,[2] the main point remains valid: north-south compensations should not omit the opportunity cost of land, measured in terms of foregone income that local economies would have to sacrifice if the current trend of land clearing is halted for global reasons.

TABLE 9.6
DISCOUNTED VALUES OF CARBON CONSERVATION OPPORTUNITY COSTS (T_0=1990)

	1990	1995	2000	2005	2010
Discount rate 2%					
Agricultural GDP (US$/ton C)	7.1	10.8	13.3	16.2	20.1
Total GDP (US$/ton C)	51.8	76.4	94.2	120.3	162.1
Discount rate 6%					
Agricultural GDP (US$/ton C)	7.1	9.4	10.0	10.5	11.2
Total GDP (US$/ton C)	51.8	66.1	70.5	77.9	90.8

On the other hand, these results do not eliminate the main motivation of north-south transfers—avoiding either the damages caused by global warming or the more expensive strategies of carbon emission abatement. Fankhauser (1994), for instance, considers a benchmark of US$ 20/t C for the social costs imposed by carbon emissions in the period 1991-2000 (ranging from US$ 6/t C to US$ 45/t C). Existing carbon taxes in Europe vary from $ 6.1/t C to US$ 45/t C (Shah and Larson 1992), and other studies present upper-limit values of more than US$ 100/t C for the 1991-2000 period (for a review, see Andersen 1996). These values clearly show that at least in a huge number of cases, total costs of carbon conservation in tropical forests (direct and opportunity costs) will be lower than the global benefits.

6. CONCLUSION

Global warming is a problem for everybody—the rich and the poor, the north and the south. Cooperation and participation aiming at the control of carbon emissions are essential, and solutions should not be restricted to a specific group of countries. Nevertheless, equity principles cannot be ignored. Therefore, the north-south agenda for cooperation should be based on fairness. There is no positive advancement if the problem of greenhouse gases is controlled at the cost of increasing disparity between developed and developing countries.

This chapter discussed the use of economic instruments (carbon taxes, offsets and tradable permits) in the control of CO_2 emissions. Economic instruments have a crucial role in an efficient cooperation between developed and developing countries. The preservation of tropical forests as carbon pools can be encouraged if the industrialized countries provide economic incentives to 'import' these environmental services which today are given for free.

However, there are important limitations to be considered. Allocation of emission rights, definition of compensation values, the possibility of free riding by non-cooperative countries, enforcement, settlement of disputes and other concrete problems are important institutional difficulties in the implementation of multilateral programmes. In spite of the considerable advances made in recent times, research of these topics remains a crucial issue.

Among them, one specific topic was developed more carefully in this chapter. To be effective, any compensation instrument should be based on proper valuation of the economic potential of the forest areas to be preserved. Nevertheless, some of the current proposals are based on the hypothesis that land prices reflect the true opportunity cost of land. One important finding of this study is that such a procedure does not respect the equity principle. The implementation of compensations based on actual land prices result in benefits to industrialized countries which may largely exceed the benefits to the countries that effectively act in the control of carbon emissions. This is shown in the exercises carried out using data for the Amazon. The risk of such a situation is that, instead of a true win-win solution, the result is a perpetuation of the wealth concentration problem on a global scale.

NOTES

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[1] A methodological description is presented in Reis (1996).

[2] Discounting is a complex issue, given the complexity of greenhouse gases discounting and the controversy about intergenerational equity.

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