Managing Forest Ecosystems: The Challenge of Climate Change

Managing Forest Ecosystems

Volume 17

Aims & Scope:

Well-managed forests and woodlands are a renewable resource, producing essential raw material with minimum waste and energy use. Rich in habitat and species diversity, forests may contribute to increased ecosystem stability. They can absorb the effects of unwanted deposition and other disturbances and protect neighbouring ecosystems by maintaining stable nutrient and energy cycles and by preventing soil degradation and erosion. They provide much-needed recreation and their continued existence contributes to stabilizing rural communities.

Forests are managed for timber production and species, habitat and process conservation. A subtle shift from *multiple-use management* to *ecosystems management* is being observed and the new ecological perspective of *multi-functional forest management* is based on the principles of ecosystem diversity, stability and elasticity, and the dynamic equilibrium of primary and secondary production.

Making full use of new technology is one of the challenges facing forest management today. Resource information must be obtained with a limited budget. This requires better timing of resource assessment activities and improved use of multiple data sources. Sound ecosystems management, like any other management activity, relies on effective forecasting and operational control.

The aim of the book series ***Managing Forest Ecosystems*** is to present state-of-the-art research results relating to the practice of forest management. Contributions are solicited from prominent authors. Each reference book, monograph or proceedings volume will be focused to deal with a specific context. Typical issues of the series are: resource assessment techniques, evaluating sustainability for even-aged and uneven-aged forests, multi-objective management, predicting forest development, optimizing forest management, biodiversity management and monitoring, risk assessment and economic analysis.

The titles published in this series are listed at the end of this volume.

Felipe Bravo • Valerie LeMay
Robert Jandl • Klaus von Gadow
Editors

Managing Forest Ecosystems: The Challenge of Climate Change

Springer

Editors
Dr. Felipe Bravo
University of Valladolid,
Palencia
Spain
fbravo@pvs.uva.es

Dr. Valerie LeMay
University of British
Columbia, Vancouver, BC,
Canada
valerie.lemay@ ubc.ca

Dr. Robert Jandl
Research and Training Centre for Forest,
Landscape and Natural Hazards (BFW),
Vienna, Austria
robert.jandl@ bfw.gv.at

Prof. Klaus von Gadow
Georg-August-Universität,
Göttingen,
Germany
kgadow@gwdg.de

ISBN 978-1-4020-8342-6 e-ISBN 978-1-4020-8343-3

Library of Congress Control Number: 2008921934

Printed on acid-free paper

9 8 7 6 5 4 3 2 1

springer.com

Foreword

Climate changes, particularly warming trends, have been recorded around the globe. For many countries, these changes in climate have become evident through insect epidemics (e.g., Mountain Pine Beetle epidemic in Western Canada, bark beetle in secondary spruce forests in Central Europe), water shortages and intense forest fires in the Mediterranean countries (e.g., 2005 droughts in Spain), and unusual storm activities (e.g., the 2004 South-East Asia Tsunami). Climate changes are expected to impact vegetation as manifested by changes in vegetation extent, migration of species, tree species composition, growth rates, and mortality. The International Panel on Climate Change (IPCC) has included discussions on how forests may be impacted, and how they may be used to mitigate the impacts of changes in climate, to possibly slow the rate of change.

This book provides current scientific information on the biological and economical impacts of climate changes in forest environments, as well as information on how forest management activities might mitigate these impacts, particularly through carbon sequestration. Case studies from a wide geographic range are presented. This information is beneficial to managers and researchers interested in climate change and impacts upon forest environments and economic activities.

This volume, which forms part of Springer's book series Managing Forest Ecosystems, presents state-of-the-art research results, visions and theories, as well as specific methods for sustainable forest management in changing climatic conditions. The book contains a wealth of information which may be useful to foresters and forest managers, politicians and the legal and policy environment and forestry administrators. The volume is subdivided into five parts. The first part presents an introduction which clarifies the context and sets the scene, in particular focusing on climatic change and its impact on forest management, the mitigation potential of sustainable forestry and the role of adaptive management and research. The second part titled "Overview of Climate Change and Forest Reponses" provide a general overview, including information about Greenhouse gas emissions from mountain forests, the capacity of forests to cope with climate change and the role of dead trees in carbon sequestration. The third part presents monitoring and modeling approaches. This includes methods to estimate carbon stocks and stock changes in forests at different scales of resolution, methods to estimate climate change impacts on forest health, an overview of forest eco-physiological models and recent advancements in techniques for assessing and monitoring carbon stocks. In the fourth part, several approaches to economic analyses of different management scenarios are presented, including the influence of carbon sequestration in an optimal set of coppice rotations, carbon in forests and wood products, and climatic impacts on forest economies, including changes in harvest cycles and the use of wood. Finally, a range of case studies on climate change impacts and mitigation activities in different ecosystems across Europe, Asia and America is presented in Part V. The case studies include forest plantations as well as tropical and Mediterranean forests.

We wish to acknowledge the valuable contributions made by our referees, for their constructive criticism and improvement. Finally, we appreciate the diligent proofreading and editing assistance provided by Celia Redondo of University of Valladolid at Palencia and Ria Kanters of Springer.

Felipe Bravo
Valerie LeMay
Robert Jandl
Klaus von Gadow

Contents

Part IV Economic and Management Impacts

Part V Case Studies

Part I
Introduction

Introduction

F. Bravo[1]*, R. Jandl[2], K. V. Gadow[3], and V. LeMay[4]

1 Forest Management and Climate Change

The recent rates of climate changes are unprecedented, given past climate change evidence. Variations in gas concentrations within the Earth's atmosphere cause changes in the climate, and these atmospheric gases are impacted by human activities. The main contributor to rates of climate change is the amount of carbon dioxide. Other gases such as nitrogen oxides and methane play a more variable role, depending on region and type of ecosystem. Current studies indicate that increases in accumulations of atmospheric gases, particularly carbon dioxide, have resulted in positive radiative forcing (i.e., the difference between incoming and outgoing radiation energy is positive) in climate systems, and this is the primary cause of global warming (IPCC, 2007; Norby et al., 2007; Raupach et al., 2007). According to these studies, the greenhouse effect, which is the warming produced as greenhouse gases trap heat, plays a key role in modifying the regulation of the earth's temperature. Projections of temperature changes and other regional-scale climatic changes, including changes in wind patterns, precipitation, and some aspects of extreme weather events, are now considered more reliable. Advances in climate change modelling now enable best estimates and *likely* uncertainty ranges to be given for projected warming and other changes for different emission scenarios (Jylhä, 2007; IPCC, 2007).

---

[1] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain, *fbravo@pvs.uva.es

[2] Research and Training Centre for Forest, Landscape and Natural Hazards (BFW), Seckendorff Gudent Weg 8, A-1131 Vienna, Austria, Robert.jandl@bfw.gv.at

[3] Retired Professor at the Faculty of Forest Sciences and Forest Ecology at the University of Göttingen in Germany, Büsgenweg 5, D-37077, Göttingen, Germany, kgadow@gwdg.de

[4] Professor with the Forest Resources Management Department, University of British Columbia, 2045–2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada, Valerie.LeMay@ubc.ca

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

Forests play a significant role in the climate system. Trees are large organisms that store carbon throughout their life and release it through decomposition. Since forests are important carbon sinks and sources, assessing forest carbon budgets has received much attention in recent years (Apps & Price, 1996; IPCC, 2000, 2001, 2007). There is a perception that humans must alter land use practices to reduce the rates of climate changes and alleviate any resulting negative social, economic, and environmental impacts. Carbon losses or gains in forests may result through afforestation, reforestation or deforestation. A recently published review of the economics of climate change stated that 18% of total annual greenhouse gas emissions today are caused by deforestation (Stern, 2006). According to Humphreys (2006), 'neoliberalism' is the key ideological force that drives excessive forest exploitation. Efforts by the World Bank to promote forest conservation are undermined by its support for the neoliberal paradigm of deregulation, privatization, and structural adjustment in indebted tropical forest countries. Humphreys stated that market-based initiatives, such as certification by the Forest Stewardship Council, may complement the public sector, but he rejected complete reliance on privatization and deplores the poor performance of UNFF[1] member governments on reporting, implementation, and failure to provide leadership and direction to other forest-related institutions.

Forest growth, structure and function are affected by the climate (Fujimori, 2001; Peñuelas et al., 2004). Increasing temperatures will cause higher respiration rates while photosynthetic rates are reduced by dry conditions. The impact of climate on a forest ecosystem will vary depending upon what factors limit tree regeneration and growth. Where low temperature is the most limiting factor, as in parts of the boreal forest biome, increasing temperatures will result in increased tree growth. However, forests under water or high temperature stress will be more susceptible to attack by pathogens and insects. In moisture-limited areas, such as the Mediterranean forests, where precipitation is expected to decrease further, tree growth will be reduced while the incidence of fire may increase.

As well as impacts on carbon balances, forests impact other atmospheric gas concentrations, water balance, biodiversity and soil stability. In areas currently subject to flood events, for example, changes in climate are expected to impact the frequency and magnitude of these events. However, the biogeochemistry of nitrogen, water yields from catchments, habitat for certain animal species are determined, not only by natural processes, but are greatly impacted by human activities. Forest management can be used to reduce the severity and extent of these climatic impacts through afforestation, preservation and conservation, alterations in harvest rates, and tree species selection, thinning regimes, and decreased reforestation delays following harvest.

---

[1] United Nations Forum on Forests.

2 Mitigation Through Adaptive Forest Management

In order to evaluate the impacts of climate change on forests, and the potential effects of management strategies to mitigate such changes, monitoring, modeling and specific research programs are needed. These activities are explicitly included into Article 3.3 of the Kyoto Protocol as “accountable activities” in the national commitments to reduce net greenhouse gas emissions (UNFCCC, 1997). “Additional human-induced activities” related to forest management of existing forests are also mentioned in Article 3.4 of the Kyoto Protocol. Such activities refer to alternative silvicultural practices which may have significant impacts on the global carbon budget (Mund & Schulze, 2006).

Forest management is a key factor in mitigating the effects of climate change. There are a number of possible strategies, including: (a) conservation and maintenance of carbon stocks which have accumulated in forests; (b) increasing carbon stocks through afforestation; (c) modification of the forest species composition and tree size distributions; (d) promoting the planting of more resilient tree genotypes; and (e) planting trees to provide shade, stabilize soils, and alter hydrology to reduce the expected impacts of precipitation and temperature changes.

In principle, forests may be considered to be carbon sinks. Therefore, increasing and conserving forest land is expected to result in less carbon being released into the atmosphere, which would slow down the rate of climate change. However, recent research indicates that forests may become carbon emitters instead of carbon sinks in the near future (Kurz & Apps, 1999; Gracia et al., 2001; Reichstein et al., 2002). Even in exceptionally hot and dry periods such as the summer of 2003 in the Northern hemisphere, productive forests can temporarily turn into C sources because climatic conditions constrain their growth (Ciais et al., 2005). This effect may be counteracted by using management practices which include better protection against forest fires, improved disease control and increasing the length of time between successive harvests (Gracia et al., 2005). For example, thinning and selection harvests to reduce tree densities could be used to reduce the probability and intensities of fires. In addition to maintaining existing forests, the area of forest land could be increased by planting trees on abandoned farm land, reducing the regeneration delay following harvests, and promoting tree planting outside forests, especially in urban areas.

Along with maintaining and augmenting the forest area, changes in forest composition may increase the amount of carbon stored. Carbon storage varies with the species and age composition of the stand, and with tree health (e.g., Bogino et al., 2006; Bravo et al., 2008). For example, Bogino et al. (2006), studied single-species stands of *Pinus sylvestris* or *Quercus pyrenaica*, and mixed stands of the two species in central Spain. They found that more carbon was stored in single-species Scots pine stands than in Pyrenean oak stands, and mixed stands had intermediate carbon levels between the two types of pure stands. Similarly, Bravo et al. (2008) found that carbon amounts in Scots pine (*Pinus sylvestris*) stands were generally higher than in comparable maritime pine (*Pinus pinaster*) stands. Such differences among species may be explained by differential growth rates, stem forms and longevity.

In the absence of forest management, species changes will occur in response to climatic changes through natural selection of better-suited genotypes, and through re-establishment of populations in more favorable climates. Trees and other plants will migrate to more suitable climates through seed dispersal and vegetative reproduction on edges of current spatial distributions. Diversity in genotypes will improve the possibility of successful adaptation to climatic changes. However, the rate of natural species adaptation is too slow to curb the impacts of the expected rapid climate change. Through forest management, however, more resilient genotypes can be planted in order to speed up the necessary adaptation. Genotypes that are drought hardy and tolerant of higher temperatures could be used where precipitation is expected to drop along with increases in temperature. In areas where shade and precipitation are expected to increase, more shade tolerant genotypes could be used. Where changes in climate are expected to affect insect and pathogen life cycles, more pest resistant trees may be introduced.

3 Monitoring and Adaptive Management

Changes in the rate and intensity of disturbance events are expected to occur as a consequence of changes in climate. Monitoring catastrophic events, such as flooding, and fire and insect damage, is common practice, but will become more important. More detailed monitoring of less catastrophic events will also be needed to indicate areas which are particularly vulnerable to regional changes in the climate.

At the same time, monitoring of the changes in carbon balance must be supported. The information provided by monitoring can be used to assess the social, economic, and environmental benefits from forests, and to provide feedback for changes to management. Remotely-sensed imagery can be used to reduce the cost of assessment. However, since tree biomass occurs both above and below ground, expensive ground sampling will be needed along with remotely sensed data.

In view of the difficulty in predicting the direction and rate of change at the local level, some of the assumptions which have guided forest management over the past two centuries must be re-examined. Forest management practices which were successful in the past may not guarantee future success and it will be necessary to re-evaluate them regularly. Nyberg (1998) proposed greater emphasis on adaptive management, which involves systematic learning on the basis of the results of past management activities. Such learning may be slow, however. For this reason, it is advisable to use new paradigms of managing forest ecosystems, together with improved modelling tools, which permit accurate forecasting and systematic evaluation of different management options based on current information about the forest resource.

One such paradigm, which is known as the Multiple Path Concept (MPC; Gadow et al., 2007) provides a suitable basis for designing forested landscapes. The mathematical formulation of the concept has been proposed and sometimes implemented in mathematical programming models in North America (Clutter et al., 1983; Hoganson & Rose, 1984; Bettinger et al., 1997) and Northern Europe (Lappi, 1992; Pukkala & Kangas, 1993;

Eid & Hobbelstad, 2000; Öhman, 2002), and has been explored for specific conditions in Central Europe (Chen & Gadow, 2002), Columbia (Schwichtenberg & Sánchez Orois, 2003) and Russia (Gurjanov & Gadow, 2005). MPC assumes that a forested landscape is an aggregation of spatially defined land parcels of varying size and shape. These geographical units are known as forest holdings, stands or compartments. They may be homogeneous or rather inhomogeneous entities, but each unit is characterized by a specific tree population with a given set of current attributes and site conditions. There is a basic understanding that not only one, but a variety of treatment schedules or "management paths" may be potentially suitable for each stand. Each stand-path is characterized by a specific succession of management activities, unexpected hazards and growth, and has a value in terms of the services that it provides. Thus, designing a forested landscape involves the search for a combination of management paths which provides a desirable mix of services to the landowner. Because carbon sequestration by forests is considered a way to reduce the rates of climate change, carbon emissions may become tradable. Therefore, an essential element of forest design is the calculation of a carbon balance for alternative management paths so that the environmental impacts can be evaluated before implementation. In accordance with the MPC-theory, the carbon balance of a forest is the sum of the carbon balances of the individual stands. The carbon balance of a particular stand during a specific time period can be calculated from the following components: (a) change of biomass, (b) mass of trees harvested during the period, (c) mass of trees which died during the period, (d) decomposition of dead trees present in the stand, and (e) decomposition of trees harvested from the stand (including roots, cutting residues and removed timber assortments; this can be calculated if the average lifespan of saw logs, pulpwood or firewood is known). The calculation of the carbon balance requires individual estimates for all these processes, rather than a single model or simple index.

4 Research

Because of the long-term environmental and social implications of forest resource management, forest research has always had to transcend boundaries. Forest scientists have joined with other disciplines, typically the biological, mathematical and social sciences, to ensure that new and specialized research results are applied to forest landscape problems. This "*integrating*" principle necessitates bridging gaps between related disciplines and incorporating their specific knowledge to provide a suitable mix of desired services. According to Sayer and Campbell (2004), this integration may not yield scientific breakthroughs, but it can help to generate options and to resolve problems.

Credible forest management is based on empirical research. The aim of early field experiments established during the 19th century was to measure timber yields on different growing sites in response to specific thinning treatments. Some of these experiments have been remeasured for over a century, providing valuable information on long-term developments (Pretzsch, 2001; Innes, 2005).

The information to be gathered in forest experiments has to be weighed against the estimated cost of data collection. Not only are the available resources limited, but time is also a major constraint. Furthermore, the validity and effectiveness of an experiment is influenced by its design and execution. Thus, attention to the planning of field experiments is important.

Gadow and Kleinn (2005) distinguish two basic types of experiments:

1. The manipulated experiment which is an investigation that establishes a particular set of circumstances under a specified protocol with the aim of testing a hypothesis
2. The comparative observational study, which involves collecting and analyzing data from different site conditions but without actively pre-defining or changing these conditions (i.e. without applying treatments) (Kuehl, 1994).

Comparative observational studies are also known as quasi-experiments (Cook & Campbell, 1979). One of the purposes for doing quasi-experimental research is to capture a sufficient number of different conditions. The intention may be to observe changes in tree growth and to attribute these changes to variables such as changes in air temperatures or carbon dioxide concentrations over time. Observational studies tend to involve many different and interacting relationships between variables; as a result, commonly, much of the variability cannot be explained by the available explanatory variables. Nevertheless, data from observational studies are often used to develop models of process and even to test hypotheses.

Field studies may be classified as *longitudinal*, *chronosequence,* or *short time series. Longitudinal studies* are long term experiments requiring high maintenance costs and a suitable research infrastructure. The object of the trial is not always achieved, as plots may be destroyed prematurely by wind or fire. Often, the original questions have become irrelevant. *Chronosequences* are measured only once, but cover a wide range of ages and growing sites. Thus, the sequence of remeasurements in time is substituted by simultaneous *point* measurements in space. Chronosequences may provide information relatively quickly, but they do not capture rates of change in response to a known initial state. A compromise may be achieved by using *short time series*, also known as *interval studies*, which maintain the advantages of *longitudinal studies* (change rates) and *chronosequences* (broad coverage of initial states and minimum decay in obtaining data). Interval field plots are measured at least twice. The interval between the measurements is sufficiently long to absorb short-term effects of climatic fluctuations.

5 Organization of This Book

The principal theme of this book is to present scientific evidence on the impacts of climate change on managed forests, and to propose forest management strategies that will mitigate the impacts of climate change. These themes will be addressed in four chapters.

Chapter 2 will give a general overview, including information about Greenhouse gas emissions from mountain forests, the capacity of forests to cope with climate change and the role of dead trees in carbon sequestration.

Chapter 3 deals with monitoring and modeling approaches. This includes methods to estimate carbon stocks and stock changes in forests at different scales of resolution, methods to estimate climate change impacts on forest health, an overview of forest eco-physiological models and sophisticated techniques of assessment and monitoring of carbon stocks.

Chapter 4 presents several approaches to economic analysis of different management scenarios, including the influence of carbon sequestration in an optimal set of coppice rotations, carbon in forests and wood products and climatic impacts on forest economies, including changes in harvest cycles and the use of wood.

A range of case studies on climate change impacts and mitigation activities in different ecosystems across Europe, Asia and the Americas is presented in Chapter 5. The case studies include forest plantations as well as tropical and Mediterranean forests.

The contributions are truly international, including authors from North and South America, Europe and Asia.

References

Apps, M.J., Price, D.T. (eds.) (1996). Forest ecosystems, forest management and the global carbon cycle. In Proceedings of the NATO Advanced Research Workshop: The Role of Global Forest Ecosystems and Forest Resource Management in the Global Cycle, Banff, Canada, September 12–16, 1994. Springer, Berlin.

Bettinger, P., Sessions, J., Boston, K. (1997). Using Tabu Search to schedule timber harvests subject to spatial wildlife goals for big game. *Ecological Modelling 94*: 111–123.

Bogino, S., Bravo, F., Herrero, C. (2006). Carbon dioxide accumulation by pure and mixed woodlands of *Pinus sylvestris* L. and *Quercus pyrenaica* Willd. in Central Mountain Range Spain. In Bravo, F. (ed.) Proceedings of the IUFRO Meeting on Managing Forest Ecosystems: The Challenges of Climate Change, Palencia, April 3–7, 2006.

Bravo, F., Bravo-Oviedo, A., Díaz-Balteiro, L. (2008). Carbon sequestration in Spanish Mediterranean forests under two management alternatives: A modeling approach. *European Journal of Forest Research* DOI 10.1007/s10342-007-0198-y.

Chen, B.W., Gadow, K.v. (2002). Timber harvest planning with spatial objectives using the method of simulated annealing. *Forstwiss. Centralblatt 121*: 25–34.

Ciais, Ph., Reichstein, M., Viovy, N., Granier, A., Ogé, J., Allard, V., Aubinet, M., Buchmann, N., Bernhofer, Chr., Carrara, A., Chevallier, F., De Noblet, N., Friend, A.D., Friedlingstein, P., Grünwald, T., Heinesch, B., Keronen, P., Knohl, A., Krinner, G., Loustau, D., Manca, G., Matteucci, G., Miglietta, F., Ourcival, J.M., Papale, D., Pilegaard, K., Rambal, S., Seufert, G., Soussana, J.F., Sanz, M.J., Schulze, E.D., Vesala, T., Valentini, R. (2005). Europe-wide reduction in primary productivity caused by the heat and drought in 2003. *Nature 437*: 529–533.

Clutter, J.L., Fortson, J.C., Pienaar, L.V., Brister, G.H., Bailey, R.L. (1983). Timber Management - A Quantitative Approach. Wiley, New York, pp. 238–252.

Cook, T., Campbell, D. (1979). Quasi-Experimental Design. Rand McNally, Chicago, IL.

Eid, T., Hobbelstad, K. (2000). AVVIRK-2000: A large-scale forestry scenario model for long-term investment, income and harvest analyses. *Scandinavian Journal of Forest Research 15*(4): 472–482.

Fujimori, T. (2001). Ecological and Silvicultural Strategies for Sustainable Forest Management. Elsevier, Amsterdam, The Netherlands.

Gadow, K.v., Kleinn, C. (2005). Forest management, science-based and understandable. In Peterson, C.E., Maguire, D.A. (eds.) Balancing Ecosystem Values – Innovative Experiments for Sustainable Forestry. US Department of Agriculture, Forest Service, Gen. Tech. Rep. PNW-GTR-635, Portland, OR, pp. 15–23.

Gadow, K.v., Kurttila, M., Leskinen, P., Leskinen, L., Nuutinen, T., Pukkala, T. (2007). Designing Forested Landscapes to Provide Multiple Services. CAB Reviews (submitted).

Gracia, C., Sabaté, S., López, B., Sánchez, A. (2001). Presente y futuro del bosque mediterráneo: Balance de carbono, gestión forestal y cambio global. In Zamora, R., Puignaire, F.I. (eds.) Aspectos funcionales de los ecosistemas mediterráneos. CSIC-AEET, Granada, pp. 351–372.

Gracia, C., Gil, L., Montero, G. (2005). Impactos sobre el sector forestal. En Moreno Rodríguez, J.M. (ed.) Evaluación preliminar de los impactos en España por efecto del cambio climático. Ministerio de Medio Ambiente, pp. 399–436.

Gurjanov, M., Gadow, K.v. (2005). Analyse der nachhaltigen Nutzung im schlagweisen Betrieb am Beispiel des Forstbetriebes Lissino bei St. Petersburg, Russland. *Forstarchiv 76*: 177–183.

Hoganson, H.M., Rose, D.W. (1984). A simulation approach for optimal timber management scheduling. *Forest Science 30*(1): 200–238.

Humphreys, D. (2006). LogJam - Deforestation and the Crisis of Global Governance. Earthscan, London, 302 pp.

Innes, J.L. (2005). Long-term forest experiments: The need to convert data into knowledge. In Peterson, C.E., Maguire, D.A. (ed) Balancing Ecosystem Values: Innovative Experiments for Sustainable Forestry. U.S.D.A. Forest Service PNW-GTR-635, pp. 25–31

IPCC (Intergovernmental Panel on Climate Change). (2000). Land use, land-use change, and forestry. Special Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

IPCC (Intergovernmental Panel on Climate Change). (2001). Climate change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York.

IPCC (Intergovernmental Panel on Climate Change). (2007). Summary for policymakers. In Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (eds.) Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Jylhä, K. (2007). What are the expected changes in climate and what uncertainties are involved? Presentation during the Seminar on Adaptive Forest Management in Changing Climate, Vantaa, Finland, March 21, 2007.

Kuehl, R.O. (1994). Statistical Principles of Research Design and Analysis. Duxbury, Belmont, CA, 686 pp.

Kurz, W.A., Apps, M.J. (1999). A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. *Ecological Applications 9*: 526–547.

Lappi, J. (1992). JLP - a linear programming package for management planning. The Finnish forest research institute. *Research Papers 414*: 1–134.

Mund, M., Schulze, E.D. (2006). Impacts of forest management on the carbon budget of European beech (*Fagus sylvatica*) forests. *Allgemeine Forst- und Jagdzeitung 177*: 47–63.

Norby, R.J., Rustad, L.E., Dukes, J.S., Ojima, D.S., Parton, W.J., Del Grosso, S.J., McMurtie, R.E., Pepper, D.A. (2007). Ecosystem responses to warming and interacting global change factors. Chapter 3, In Canadell, J.G., Pataki, E.E., Pitelka, L.F. (eds.) Terrestrial Ecosystems in a Changing World. The IGBP Series, Springer, Heidelberg.

Nyberg, J.B. (1998). Statistics and the practice of adaptive management. In Sit, V., Taylor, B. (eds.) Statistical Methods for Adaptive Management Studies. Research Branch, British Columbia Ministry of Forests, Land Management Handbook No. 42, Victoria, British Columbia, Canada, pp. 1–7.

Öhman, K. (2002). Forest planning with consideration to spatial relationships. Doctoral dissertation. Department of Forest Resource Management and Geomatics, SLU. Acta Universitatis agriculturae Sueciae, Silvestria, Volume 198.

Peñuelas, J., Filella, I., Zhang, XY., Llorens, L., Ogaya, R. (2004). Complex spatiotemporal phenological shifts as a response to rainfall changes. *New Phytologist 161*(3): 837–846.

Pretzsch, H. (2001). Modellierung des Waldwachstums. Parey Buchverlag, Berlin, 341 S.

Pukkala, T., Kangas, J. (1993). A heuristic optimization method for forest planning and decision making. *Scandinavian Journal of Forest Research 8*: 560–570.

Raupach, M.R., Marland, G., Ciais, P., Le Quéré, C., Canadell, J.P., Klepper, G., Field, C.B. (2007). Global and regional drivers of accelerating CO_2 emissions. *Proceedings of the National Academy of Sciences, USA 104*(24): 10288–10293.

Reichstein, M., Tenhunen, J.D., Roupsard, O., Ourcival, J.M., Rambal, S., Miglietta, F., Peressotti, A., Pecchiari, M., Tirone, G., Valentini, R. (2002). Severe drought effects on ecosystem CO_2 and H_2 O fluxes at three Mediterranean evergreen sites: Revision of current hypothesis? *Global Change Biology 8*: 999–1017.

Sayer, J., Campbell, B. (2004). The Science of Sustainable Development. Cambridge University Press, Cambridge.

Schwichtenberg, G., Sánchez Orois, S. (2003). Optimising management alternatives in an Acacia-mangium-plantation in Columbia. (In Spanish). In Proceedings of the VI Congreso Mexicano de Recursos Forestales, San Luís de Potosí, Mexico.

Stern, N. (2006) The Economics of Climate Change. The Stern Review. Cambridge University Press, Cambridge, 692 pp. http://www.hm-treasury.gov.uk/independent_reviews/stern_review_economics_climate_change/sternreview_summary.cfm, last accessed November 2007.

UNFCCC (United Nations Framework Convention on Climate Change). (1997). The Kyoto Protocol. http://unfccc.int/kyoto_protocol/items/2830.php, accessed October 2007.

Part II
Overview of Climate Change and Forest Responses

A Mechanistic View of the Capacity of Forests to Cope with Climate Change

F. Valladares

1 Introduction

From an evolutionary point of view, trees have at least one intriguing feature: they tend to have high levels of genetic diversity, but at the same time, they are known for their low evolutionary rates. Thus, trees are characterized by a counterintuitive combination of rapid micro-evolutionary change and a low macro-evolutionary change (Petit & Hampe, 2006). Trees experience highly heterogeneous environmental conditions and are exposed to extreme climatic events within their lifetime, which could contribute to the maintenance of their typically high genetic diversity (Gutschick & BassiriRad, 2003; Petit & Hampe, 2006). Trees are not only highly diverse but also highly fecund over their extended lifetime, allowing them to respond to high selection intensity and to adapt quickly to local conditions (Petit & Hampe, 2006). Mean antiquity of tree species is one order of magnitude higher than for herbs, which implies low rates of extinction to compensate for their low rates of speciation. However, forest species are more vulnerable to environmental change than this combination of evolutionary features may suggest (Jump & Peñuelas, 2005). Recent studies of Spanish populations of beech (*Fagus sylvatica*) are showing that the fragmentation of the forests that took place several centuries ago has led to a high genetic divergence of the populations and a reduced genetic diversity despite the fact that the species is wind-pollinated and the fragments are very near to each other (Jump & Peñuelas, 2006). These studies show the negative genetic impact of forest fragmentation, demonstrating that trees are not at reduced risk from environmental change (Fig. 1). This rather unexpected sensitivity of trees to forest management is particularly important under the current climate change since it can exacerbate the impact of human activities on forest dynamics and natural regeneration (Castro et al., 2004a).

The capacity of forests to cope with climate change has been considered to be relatively ample, and many physiological, genetic and evolutionary aspects have been suggested to contribute to the persistence of key forest trees and plants in a changing climate. However, the fast rate of current environmental change is imposing severe

---

Instituto de Recursos Naturales. CSIC, IRN-CCMA-CSIC, Serrano 115, E-28006 Madrid Spain,
Fax 34 915640800 ext 1, 204, valladares@ccma.csic.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

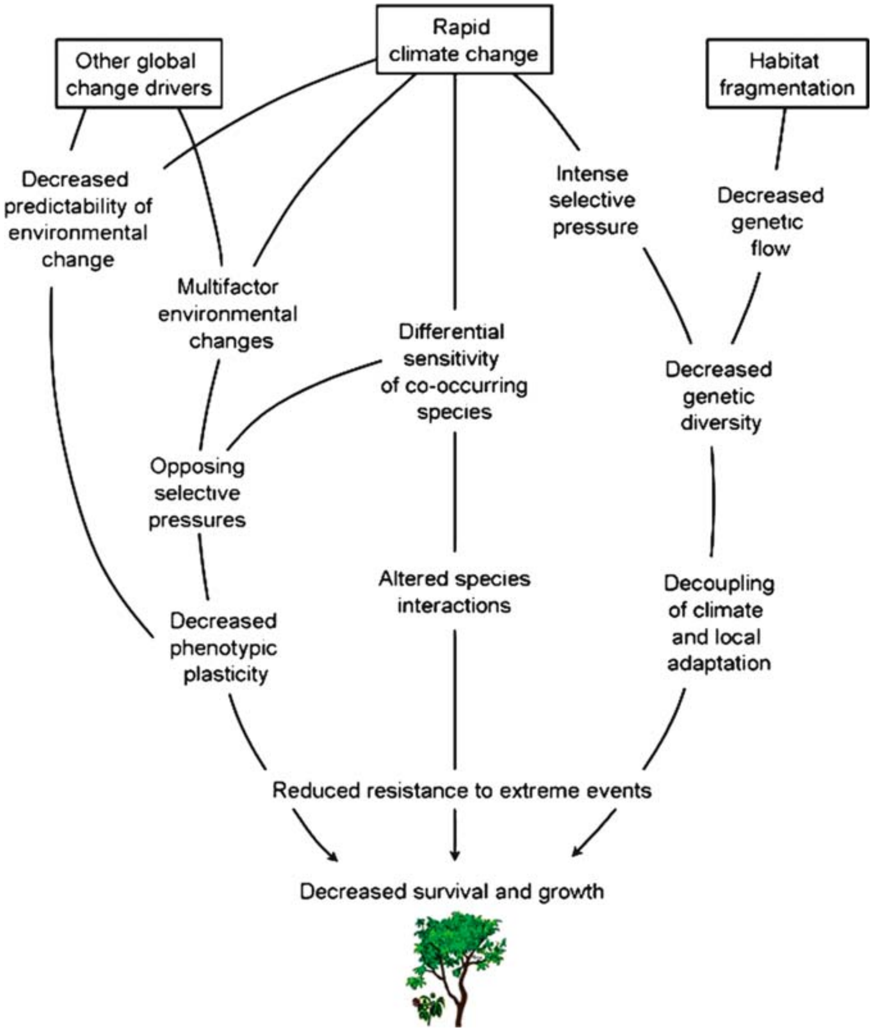


Fig. 1 Interactive effects of global change drivers on plant survival and growth. Rapid climate change is at the centre of a whole suite of environmental changes that is imposing complex and opposing selective pressures to forest species; this in turn limits the extent and ecological benefits of phenotypic plasticity, modifies species interactions, and decouples climate and local adaptation, leading to an increased vulnerability to extreme climatic events and to a higher risk of mortality under the new climatic scenarios. The ideas of habitat fragmentation and climate change effects on genetic diversity and local adaptation are taken from Jump and Peñuelas (2005)

limitations to the capacity of trees to adapt to new climatic conditions (Alcamo et al., 2007). For example, levels of heritable variation for date of budburst, a crucial plant trait involved in the responses to global warming, were considerable but inadequate to track forecast changes in climate in two *Betula* species (Billington & Pelham, 1991), and similar results were obtained for variation of bud set and frost hardiness in *Pinus sylvestris* (Savolainen et al., 2004). It must be taken into account that adaptation to future climates may require the simultaneous evolution of a number of different traits, which is constrained by correlations between them as discussed by Jump and Peñuelas (2005). Besides, climate change is only one environmental challenge directly or indirectly imposed to natural ecosystems by human activities, while it is the combined effect of climate change with loss and fragmentation of habitats, loss of soil, pollution and introduction of exotic species that is significantly reducing the regeneration and long term survival of many forest species (Valladares, 2004b). And forests are far more complex than just the sum of a given number of individual trees. Biotic interactions among co-occurring plants, animals and microorganisms are considered crucial for ecosystem functioning but our understanding of them and of their sensitivity to global change is very limited (Bascompte et al., 2006; Peñuelas & Filella, 2001). The simple fact that not all species are equally sensitive to global change leads to the realization that global change can have greater and more complex effects on communities than on individual species.

A look into human history reveals that human induced deforestation and environmental degradation coupled with climate change has led to the collapse of civilizations as developed and rich as Maya and Anazasi (Diamond, 2005). Our current civilization shares many circumstances with old civilizations that disappeared due to overexploitation of natural resources under unfavourable climatic conditions, but has a number of unique features that could prevent its collapse, namely a sophisticated technology, a rapid transfer of information and a global view of environmental problems. Understanding the limits of natural systems to cope with multifactor environmental changes can potentially improve our capacity to preserve them and to manage them in a sustainable way. I wrote this chapter with this hope in mind.

2 The Complexity of Climate Change and of Its Effects

2.1 The Many Sides of Climate Change

Climate is changing rapidly and in various and complex ways since anthropogenic climate change is much more than global warming. It is not only mean air temperatures that are rising but also the frequency of extreme climatic events (Meehl et al., 2007). Unusual heat waves and frosts are becoming more frequent together with severe droughts in arid and Mediterranean regions and floodings in many temperate and subtropical zones (Castro et al., 2004b; Christensen et al., 2007; Inouye, 2000). The intensity of the radiation reaching the ecosystems is also changing. After studies reporting a global dimming (i.e. a global reduction in the radiation reaching the ecosystems due to reductions in the transmittance of the atmosphere, Stanhill & Cohen, 2001) during the second half of the twentieth century, recent revisions report a widespread brightening, showing a reversal from dimming to global brightening during the 1990s in agreement with changes in cloudiness and atmospheric transmission (Pinker et al., 2005; Wild et al., 2005). Irradiance is, thus, globally changing, with contrasting trends towards dimming or brightening depending on the particular region of the world (Matesanz et al., 2007; Wild et al., 2005), and with a relative increase in diffuse over direct radiation due to increasing cloudiness and density of atmospheric particles (Roderick et al., 2001; Stanhill & Cohen, 2001). All these climatic changes associated with global warming are leading to significant changes in the energy and mass balances in ecosystems all over the Earth. Obviously, underlying most of these climatic alterations is the rising of the concentration of greenhouse gases, particularly of carbon dioxide. And carbon dioxide is relevant not only because it is the key greenhouse gas but also because it is the very substrate of photosynthesis so plant productivity and vegetation dynamics are directly and indirectly influenced by atmospheric concentrations of CO_2 (Fig. 2). Thus, climate change is in reality a complex mix of changes in frequency and intensity of a wide range of factors. The different components of the current climate change affect differently each hierarchical level of the ecosystem, leading to cascade effects and complex feedbacks when responses are analyzed in processes ranging from the molecule to the whole ecosystem (Fig. 2).

2.2 The Many Sides (and Scales) of Ecosystem Responses to Climate Change

Terrestrial ecosystems exposed to the many aspects of climate change are already showing effects and responding (Buchmann, 2002; Camarero & Gutiérrez, 2004; Dullinger et al., 2004; Richardson et al., 2006; Saxe et al., 2001; Menzel et al., 2006). Perhaps one of the most evident and general ecosystem effects of climate change is an altered energy and mass flux, and, in particular, a modified rate of evapotranspiration. Counterintuitively, evaporative demand from atmosphere (i.e. pantranspiration, which is correlated with potential evapotranspiration) is globally declining despite the rise of temperatures. The trend for decreasing evaporative demand has been reported throughout the Northern Hemisphere terrestrial surface and it seems to be also widespread in the Southern Hemisphere, as part of a greenhouse-related phenomenon (Roderick & Farquhar, 2005). Cloudiness and decreased wind are the main reasons argued to explain this global pattern, but the real causes of this unexpected trend are far from established. Global warming is expected to increase evapotranspiration, but experimental studies of plant communities rendered a more complex picture (Zavaleta et al., 2003). Another surprising trend in terrestrial ecosystems exposed to climate change is the increase in continental runoff through the twentieth century despite the more intensive human water consumption (Gedney et al., 2006). Climate change and variability, deforestation, changes in irradiance and direct atmospheric CO_2 effects on plant transpiration have been suggested as possible reasons for this globally increased runoff. Using a mechanistic landsurface model and optimal fingerprinting statistical techniques to

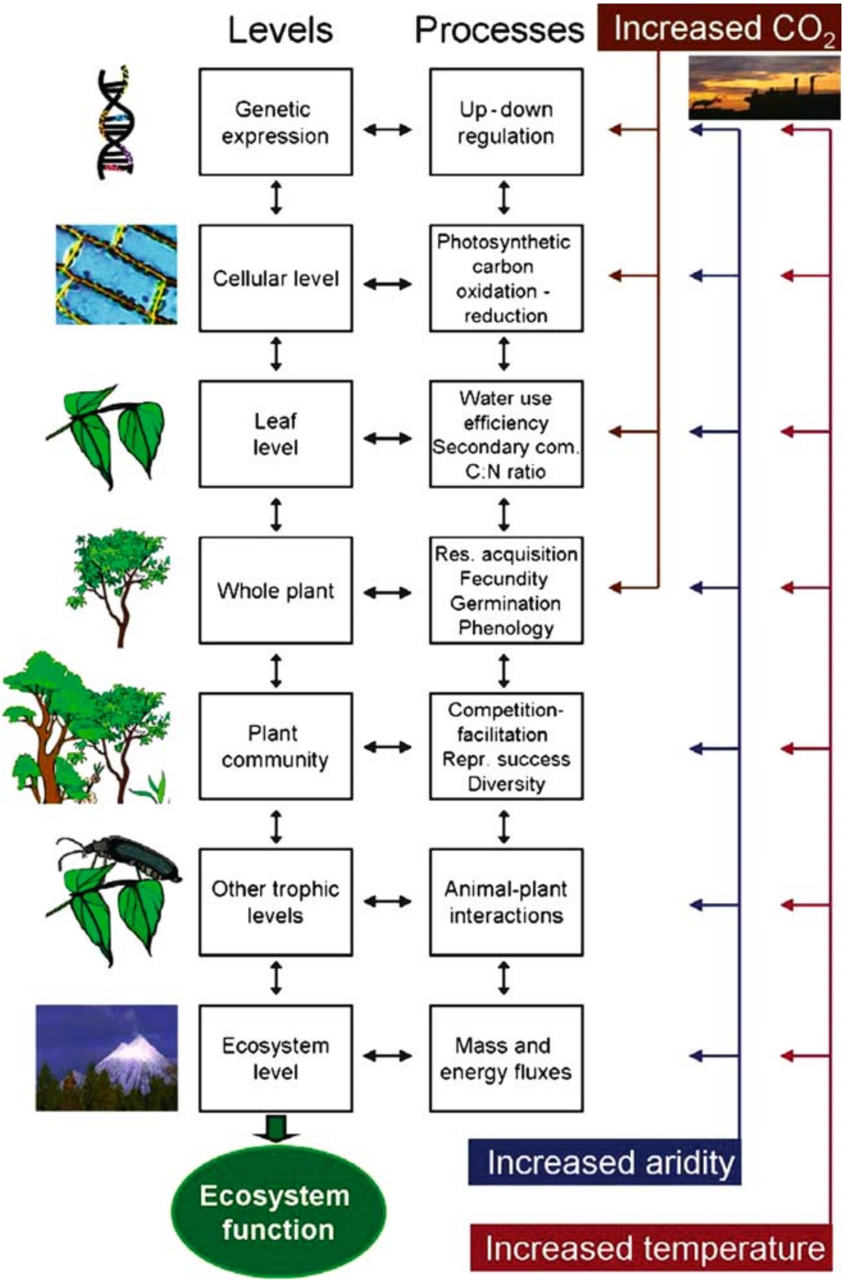


Fig. 2 Impacts of rising atmospheric CO_2 concentrations and the associated increases in temperature and aridity for different organizational levels from the molecule to the ecosystem. Main processes being affected at each level are indicated. Inspired on ideas by Ziska and Bunce (2006) on plant responses to rising CO_2

attribute observational runoff changes to these factors, it was concluded that twentieth-century climate alone is insufficient to explain the runoff trends. The trends were consistent with a suppression of plant transpiration due to CO_2-induced stomatal closure, representing the detection of a key direct CO_2 effect on the functioning of the terrestrial biosphere (Gedney et al., 2006).

All this illustrates well that ecosystem responses to climate change are different at different spatial and temporal scales. There is a globally decreased evapotranspiration and increased runoff at a very large spatial scale, with important heterogeneities at intermediate scales, such as in arid and Mediterranean regions that tend to exhibit the reverse pattern. These heterogeneities can be magnified at the local or micro spatial scale, so evapotranspiration can decrease in a region of increasing aridity by counterintuitive responses of the vegetation. And the same applies to temporal scales, so results have to be interpreted differently if responses are explored within decades (e.g. Martínez-Alonso et al., 2007), between decades (e.g. Matesanz et al., 2007), over centuries (e.g. Woodward, 1987) or over longer periods of time. It must be noted that the notion of what is long for a period of time is logically dependent on the ecosystem process or property of interest (for a thorough discussion and interesting examples see Greenland et al., 2003).

2.3 The Complexities Underlying Basic Responses to Climate Change

Climate has a strong control on plant survival, growth and reproduction. And vegetation is not only responding to warming and changes in water availability but also to changes in the diffuse fraction of irradiance, so its productivity and structure is strongly influenced by changing clouds and atmospheric particles. Thus, changes in irradiance are both cause and consequence of climate change, and direct effects on vegetation and feedbacks are complex but significant. As an evidence of this, the decline in atmospheric CO_2 concentration observed following the mount Pinatubo eruption was in part caused by the increased vegetation CO_2 uptake induced by the enhanced diffuse fraction due to volcanic aerosols (Roderick et al., 2001). Quantity and quality of the irradiance in the understory of a forest or within its canopy is crucial to many aspects determining ecosystem functioning including not only productivity but also species interactions and dynamics (Valladares, 2003, 2004a).

A changing climate is leading to a changing distribution range of plants and animals (Alcamo et al., 2007; Kullman, 2002; Parmesan, 1996; Sturm et al., 2001). Climate change-induced changes in forest growth and distribution are the subject of intense investigation because of their impacts on the terrestrial carbon sink (Saxe et al., 2001). And climate change is leading not only to distribution shifts but also to phenological shifts, with significant advances in the timing of leafing and delays in the timing of leaf shedding, combined with changing reproductive and productivity peaks in many plant species (Peñuelas & Filella, 2001; Richardson et al., 2006; Menzel et al., 2006; Alcamo et al., 2007). Seasonal cycles will be differentially affected by climate change since species not only differ in their sensitivity to environmental changes but also in the cue that triggers their response. For instance, budburst is triggered by either warm temperatures, longer days or both depending on the species, and since only temperature but not daylength is changing only some species will anticipate budburst as climate change progresses (Sanz-Perez et al., 2008). These altered phenologies coupled with differential tolerances of co-occurring species to distorted temperatures and water availabilities are leading to quick changes in the competitive abilities of species (Ogaya & Peñuelas, 2003; Peñuelas & Filella, 2001). But the overall result in terms of forest regeneration and dynamics is unknown. The few experimental and realistic results on climate change effects on community composition revealed that warm temperatures and drought resembling extreme climatic scenarios make the new assemblage of plant communities unpredictable, with composition and abundance changes affecting both common and rare species (Lloret et al., 2004).

Despite the capacity of individual species to modify their phenology and to respond to climate change by acclimation and phenotypic plasticity, the overall performance of forest species seems to be, in general, negatively affected by climate change (Alcamo et al., 2007). For instance, and contrary to expectations, ecosystem water-use efficiency of photosynthetic carbon uptake decreased during an exceptional drought in three Mediterranean forests dominated by Holm oak (Reichstein et al., 2002). Populations of beech at their lower and southern most ranges are growing less (e.g. beech forests in Montseny, Spain, annual secondary growth is 49% less now than 50 years ago when mean temperature was 1.65°C lower, and the associated evapotranspiration was significantly lower as well; Jump et al., 2006). Tertiary relict populations of *Frangula alnus*, *Rhododendron ponticum*, and *Prunus lusitanica*, among other species, are having difficulties in rendering viable seeds and juveniles in Mediterranean habitats of increasing aridity (Hampe, 2005; Mejías et al., 2002; Pulido et al., 2007).

2.4 Many Approaches to One Elusive Goal: A Mechanistic Understanding of the Responses

There is a globally increasing interest in monitoring and understanding the responses of Earth ecosystems to climate change (Alonso & Valladares, 2007), and there are many approaches to the study of forest responses to this complex environmental threat (Nabuurs et al., 2007; Valladares, 2004b). Many important efforts of national and international research programs have focused on basic ecophysiological studies of forests aimed at monitoring and understanding their net gas exchange (e.g. Morales et al., 2005). These research programs have rapidly scaled up in technological sophistication and ambition but they have remained surprisingly similar to the classical plant ecophysiology studies of the 1970s (Buchmann, 2002). Even though this research is essential for carbon balance modelling and for calculating the complex mass and energy balances at the biosphere level, I argue that they are insufficient not only to accurately estimate forest growth under future climatic scenarios but also to fully account for the already observed responses of forests to global change. As I will develop in the following, there are both important ecophysiological uncertainties that can alter the calculations of whole ecosystem carbon and water balances, and poorly understood ecological and evolutionary aspects that can significantly affect the response of forest ecosystems to global change.

3 Climate Change and Tree Physiology

3.1 Carbon Dioxide, at the Origin of the Problem and at the Core of Mitigation Plans

Climate change is primarily induced by increased greenhouse concentration in the atmosphere, with CO_2 as the most important one. But as mentioned before, CO_2 has itself an effect on plant performance since it is the basis for photosynthesis and growth and it has a significant influence in stomatal opening (Fig. 2, Gedney et al., 2006; Fig. 2, Lambers et al., 1998). It has been assumed that because increases in atmospheric CO_2 concentration usually enhance water use efficiency per unit leaf area, there will be a tendency for plants to show greater drought tolerance as well as increased biomass in the future. But critical examinations of plant responses to elevated CO_2 show that this assumption is seldom correct (Beerling et al., 1996; Körner, 2003a).The progressive increase in the concentration of atmospheric CO_2 over the past centuries might have accentuated differences in drought sensitivity between co-occurring tree species but does not seem to have led to a generally increased water use efficiency and growth (Beerling et al., 1996). General revisions of available information reveal that plant growth does not seem to be limited by carbon supply in a range of contrasting habitats, suggesting that little if any leeway exists for further CO_2 fertilization effects on growth (Körner, 2003a).

Forest ecophysiology has gained increased recognition due to the potential insights for understanding and managing terrestrial carbon sinks (Grace, 2004). Carbon sinks develop in ecosystems that have high carbon storage, such as mature forests, when these systems increase productivity, so that carbon gains by photosynthesis run ahead of carbon losses by respiration, and the stocks of carbon therefore increase (Grace & Zhang, 2006). The required stimulation may occur through elevated CO_2 concentration, nitrogen deposition or by climate change. Sinks also occur during the 'building' phase of high carbon ecosystems and there is agreement on the fact that carbon sinks are important in tropical, temperate and boreal forests, although their effect on a global scale is largely offset by deforestation in the tropics (Grace, 2004). Unfortunately, although the Kyoto Protocol provides incentives for the establishment of sinks, it does not provide incentives to protect existing mature ecosystems which constitute both stocks of carbon and carbon sinks (Grace, 2004).

Since respiration rates scale more rapidly with temperature than photosynthetic rates, understanding the effect of temperature on plant respiration is fundamental to predicting the impact of global change on the biosphere (Atkin & Tjoelker, 2003; Zaragoza-Castells et al., 2007b). But respiration is not as well understood as photosynthesis (Cannell & Thornley, 2000; Grace & Zhang, 2006). Although respiration has been shown to be very sensitive to short-term changes in temperature (i.e. exponential rise with temperature with a Q_{10} of 2), the impact of long-term temperature changes depends on the degree of respiratory acclimation, which is not accounted for in most models (i.e. Q_{10} is not constant Atkin et al., 2006; Atkin & Tjoelker, 2003). Respiration, which is an important component of carbon exchange in most terrestrial ecosystems and can release more than half of the total carbon fixed by photosynthesis, becomes particularly relevant in low-productivity ecosystems such as Mediterranean and boreal evergreen forests, since minor changes in respiratory rates may change the very sign of the overall carbon balance of plants living under limiting conditions and, in turn, of the whole ecosystem (Zaragoza-Castells et al., 2007a). In these ecosystems, the reduced photosynthesis and increased respiration associated with climate change might increase the frequency and length of periods of negative carbon balance as suggested by studies in Mediterranean Holm oak forests (Joffre et al., 2001; Rambal et al., 2004; Zaragoza-Castells et al., 2007a).

During the last decade of the twentieth century, deforestation in the tropics and forest regrowth in the temperate zone and parts of the boreal zone were the major factors responsible for CO_2 emissions and removals, respectively (Barker et al., 2007). However, the extent to which the loss of carbon due to tropical deforestation is offset by expanding forest areas and accumulating woody biomass in the boreal and temperate zones is an area of contention since actual land observations and estimates using top-down models do not match. The growing understanding of the complexity of the effects of land-surface change on the climate system shows the importance of considering the role of surface albedo, the fluxes of sensible and latent heat, evaporation and other factors in formulating policy for climate change mitigation in the forest sector. Complex modelling tools are still to be developed to fully consider the climatic effect of changing land surface and to manage carbon stocks in the biosphere. The potential effect of projected climate change on the net carbon balance in forests, thus, remains uncertain (Barker et al., 2007).

3.2 Warming Temperatures and the Short Term Dimension of Our Ecophysiological Knowledge

One of the main problems of understanding the effects of climate change on plant performance derives from the short-term nature of most ecophysiological studies. While most ecophysiological processes exhibit a dramatic response to a sudden increase in temperature, many regulatory and feedback mechanisms, and the capacity of plant to acclimate, significantly reduce the extent of such a response over the long run. This is exemplified by Körner (2006) with two paradoxes: (i) the high contrast in the productivity of ecosystems operating at contrasting temperatures vanishes when the productivity is divided by the number of months available for growth, although the case is valid only for native vegetation with no severe water limitations; (ii) while soil metabolism is very sensitive to temperature, respiratory fluxes during the growing season is quite similar from the Arctic to the tropics, being substrate driven and not temperature driven. The take home message from these two paradoxes is that temperature differences more than five times larger than those expected in the worse climate change scenario can have almost no effect on key ecosystem processes provided that there is time enough for the ecosystems to adapt to the new conditions. From a mechanistic point of view, the main challenge of current climate change for forest ecosystems is thus not the magnitude of the temperature rise but the speed of the rising together with the co-occurrence of many other environmental changes.

The time dimension of the study and of the plant responses to changing conditions have an important bearing on the choice of the response variable. While many ecophysiological efforts have gone into the characterization of photosynthetic responses, which are intrinsically short-termed, much less research has focused on plant growth, an essential ingredient to understand whole plant performance that is long-term. And the available studies show a remarkable uncoupling between growth and photosynthesis due to the important influence on growth of tissue density and duration and whole plant allometry (Körner, 2006). The scarcity of sound studies determining growth rates in a range of conditions should move more research in this direction if we are to understand plant responses to climate change since growth data are more informative than photosynthesis data: actual photosynthetic carbon gain is much less sensitive to temperature than growth, and growth is more strongly related to plant life than photosynthesis (Körner, 2006).

3.3 Climate Change as a Source of Stress

Ecophysiologists have been always attracted by the study of plant tolerance and responses to stress (see for example Larcher, 1995). And climate change is bringing a whole suite of abiotic stresses such as extreme temperatures, excessive irradiance and increased aridity, which are classic targets of ecophysiological studies. We now know that changing extreme temperature events are more relevant for plant survival than changing mean temperatures, with low-temperature extremes being particularly important. The climate is getting warmer but the chances for late or early season frosts are also increasing (Christensen et al., 2007; Meehl et al., 2007). The dangerous periods for plants are not the coldest or the hottest moments of the year, but the transitions, when the extreme event hits plants that are either dehardened or not fully hardened (Taschler & Neuner, 2004). And these transitional periods are getting less predictable and more variable. There is a common misconception that plants from cold habitats are cold stressed while they are in fact stressed when temperatures rise (Körner, 2003b). Global warming is favouring the invasion of cold habitats by frost sensitive species, which are outcompeting native, cold-adapted plants. But unusual frost events are then killing these invaders and the net result is a loss of species and a malfunctioning of the whole ecosystem. Climate change is thus challenging the very concept of stress and it is opening new avenues for research on stress physiology.

3.4 Our Limited Understanding of Co-occurring and Interacting Stresses

Plants under natural conditions are simultaneously exposed to many limiting factors, and climate change is making this combination far more complex and intense. Although increasing attention is being paid to responses to multiple stresses, most of our knowledge comes from studies on responses to single stresses (see discussions in Mooney et al., 1991; Valladares & Pearcy, 1997). Recent research has shown that the response of plants to a combination of several abiotic stresses is unique and cannot be directly estimated from plant responses to each of the different stresses applied individually (Mittler, 2006). Thus, the main ecophysiological challenge now relies in understanding plant responses to complex stresses (e.g. late frost, where timing and duration is even more important than temperature), to several interactive stresses (e.g. high light and drought or high light and freezing temperatures) and to relatively new combinations of stresses (e.g. low light and drought, Fig. 3), without overlooking biotic stresses (e.g. those induced by competing neighbours, herbivores, pathogens etc.; Figs. 3 and 4). This is not only a more attractive research arena but also an approach more likely to give a realistic view of ecosystem responses to global change, a truly multifactor phenomenon.

The effects that below-freezing temperature (frost) can have at times of year when it is unusual are an interesting ecological phenomenon that has received little attention (Inouye, 2000). The degree to which plants will suffer from frost damage in the future will depend on the interactions between temperature and precipitation, both of which are predicted to change, and also on the timing of cold snaps. According to some models, alder, birch and poplar trees will be affected and in general early-flowering trees will suffer greater frost damage in the future (Cannell et al., 1989; Howe et al., 2003; Inouye, 2000). The potential for climate change to influence the frequency and distribution of frost events is not fully understood yet but it is clear that will be very different for different regions, becoming more frequent in some areas and less frequent in others (Christensen et al., 2007). Since the impact of frost events are also very dependent on the microclimatic and microtopographic circumstances of each site, and also on the frequency and duration of the events, they are a good example of a complex climate-change related phenomenon resulting from the interplay of different temporal and spatial scales.

A canopy imposes some light limitations to understory plants but it provides some protection against excessive irradiance, particularly harmful when photosynthesis is impaired by temperature or lack of water (Valladares et al., 2005a),

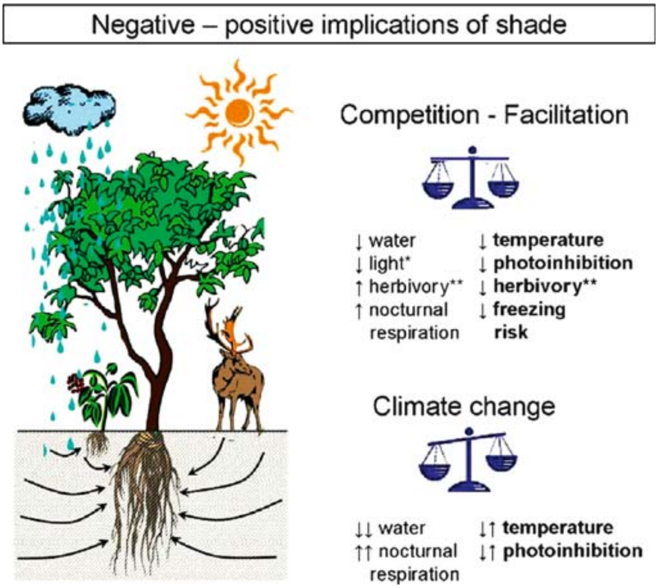


Fig. 3 Many forest plants get established in the shade due to either higher density of propagules or better environmental conditions than in open microsites. However, not all effects of established plants (i.e. those casting the shade) are beneficial for understory plants. There is a balance of negative and positive implications of being in the shade. Understory plants compete with established plants for water and light and the latter affect herbivory and nocturnal respiration (warmer overnight temperatures) of the former. Understory plants may obtain some benefits by alleviated photoinhibition and extreme temperatures, although light availability might be too low in the shade. Even though transpiration is reduced in the shade, the overstory reduces the amount of precipitation reaching the ground and the roots of the established plants might deplete most of the water in the soil, leading to a rather dry shade. Climate change is expected to increase the negative effects of the shadein dryland ecosystems by further reducing water availability and by increasing nocturnal respiration of understory plants. Climate change is likely to decrease the magnitude of the positive effects of the shade in a number of ecological situations. * the negative impact of low light on carbon gain and growth depends on the shade tolerance of the understory plant (protégée); ** herbivory has been shown to be higher in the shade since many animals spend more time feeding in the shade and plants tend to be less protected against herbivory when growing in the shade, but some nurse plants can bring special protection against herbivores, so the balance between the positive and negative impacts of shade on herbivory damage depends not only on the pressure and ecology of the herbivores but also on nurse plant identity

and it also protects against radiation frost (Ball et al., 1991). Thus, the shade represents a fairly balanced situation of positive and negative aspects, with the outcome depending on the ecological and physiological features of each species occurring in the understory (Fig. 3). I argue that this balance is affected by climate change by making the negative aspects relatively more important than the beneficial ones, at least for plants from ecosystems characterized by a short growing season such as Mediterranean and arid ones.

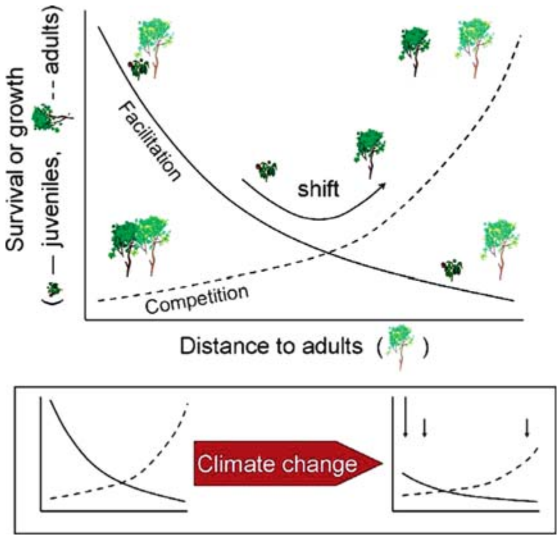


Fig. 4 Performance of juvenile plants (expressed by any surrogate of fitness like survival or growth) versus distance to adult, established plants in the field. The initial facilitation of juveniles by adult plants shifts into competition at a certain ontogenetic stage (indicated by the arrow in the upper graph), so an increasing distance to adults becomes progressively more beneficial as the target plant grows. This ontogenetic shift may help to reconcile contrasting results on facilitation vs. competition outcomes on plant-plant interaction in dry ecosystems. It must be noted that the distance to the adult plant required for the shift from facilitation to competition is relative to the size of the adult plant; in the case of shrubs, the transition can be difficult to detect since nurse and protégé compete at early stages and short distances so the balance of the interaction may fluctuate more over time and over ranges of environmental conditions than in the case of trees; however, competition between understory plants and trees is highly asymmetrical so positive interactions might be less frequent. Plant performance will be in general negatively affected by climate change, and facilitation of juveniles by established adults is expected to be particularly reduced due to earlier and more pronounced competition for water. The idea of the ontogenetic shift in the facilitation by neighbour plants is taken from Miriti (2006)

Plant ecophysiology in the shade is particularly important for a mechanistic approach to the response of forest to climate change because on the one hand most of the individuals and species that will make up the forest of the future get established in the shade, and, on the other hand, several potentially limiting or stressful factors affected by climate change co-occur in the shade (Gómez-Aparicio et al., 2006). Low light interacts with altered water and nutrient availabilities, temperature and herbivore pressure among other factors (Fig. 3). Shade is in itself a stress for most plants (Valladares & Niinemets, 2008), and it has been shown that shade tolerance is decreased under dry conditions, with significantly different responses among species to the interaction between low light and limited water availability (Sánchez-Gómez et al., 2006a, b).

Nocturnal temperatures in the shade are relatively warmer than in the open due to decreased radiative and convective heat losses; nocturnal air temperature in the shade can be up to 2.5°C higher than in the open as has been found in continental Mediterranean forests (Valladares et al., 2007b). The impacts of these warmer nights, expected to become even warmer, on the carbon balance of understory plants requires attention, particularly in low productivity forests where minor changes in respiration rates can affect the net balance of carbon. Temperature response of photosynthesis interacts with light availability, so the lower the light, the lower the temperature sensitivity of photosynthesis (Körner, 2006). Photosynthesis can be said to be largely driven by light with temperature playing only a marginal role. But this is not the case of respiration, so the warmer nocturnal temperatures experienced in the shade can have an important negative impact on the carbon balance of understory plants due to an enhanced respiratory carbon release in plants that are already carbon-limited for being in the shade (Fig. 3). However, not always the consideration of additional factors brings in more complexity: a recent study reveals that predictive carbon cycle models can assume that growth irradiance and photosynthesis do not significantly affect the temperature sensitivity of respiration of long-lived evergreen leaves (Zaragoza-Castells et al., 2007b).

Our understanding of plants coping with stress is further limited by two facts: (i) stress tolerance changes over the ontogeny of the plant (Niinemets, 2006), which is of great importance for long-living species such as trees, and (ii) stress tolerance is not achieved by a single combination of traits or trait values, so evolutionary processes and individual responses to ecological conditions do not necessarily match as has been shown for tolerance to drought (Valladares & Sánchez-Gómez, 2006). The natural occurrence of simultaneous gradients of multiple abiotic factors (light, temperature, water and fertility) makes polytolerance, i.e. the capacity to withstand two or more uncorrelated stress factors, highly adaptive. However, the empirical evidence and a recent revision of stress tolerance of Northern Hemisphere trees and shrubs suggest that it is hard to achieve if possible at all (Niinemets & Valladares, 2006).

4 Climate Change, Phenotypic Plasticity and Rapid Evolution

All organisms exhibit a certain degree of ‘phenotypic plasticity’, which is the ability of individuals to modify their behaviour, morphology or physiology in response to altered environmental conditions. And it has been suggested many times that plasticity is an effective way to cope with environmental change in general and with climate change in particular (see references in Valladares et al., 2006). Plasticity *sensu lato* includes all sort of phenotypic responses to the environment that take place at different time scales, that might or might be not reversible, and that might or might be not adaptive (Piersma & Drent, 2003). In a rapidly changing environment, narrowly adapted populations with low plasticity in important characters might involve high probability of extinction. However, little is known about the plasticity of many key plants, particularly of those of great longevity such as trees (Rehfeldt et al., 2001; Valladares et al., 2005a).

Various studies argue that global change should in principle favor high levels of phenotypic plasticity in plants (e.g. Rehfeldt et al., 2001; Parmesan, 2006). But more often than not, global change involves simultaneous changes in two or more abiotic and biotic factors, which can be expected to impose restrictions on plastic responses to the environment. Different pieces of evidence suggest that there is not a universal plasticity level that enhances fitness under multifactor changes. The complexity emerging from the simultaneous effects of several species and factors together with the interactions among them can explain the coexistence of species with contrasting plasticities and questions the notion that plastic phenotypic responses to global change are always adaptive (Valladares et al., 2005b). Global change might alter phenotypic integration as suggested by the uncoupling of growth, foliage dynamics and cone production induced by mid-term climatic variability in a Scots pine population at its southern range (Martínez-Alonso et al., 2007). Thus, global change may both induce differential plastic responses in co-occurring species and influence features such as phenotypic integration that may in turn influence plasticity for certain traits (Valladares et al., 2007a).

As argued for the tolerance to multiple stresses, plasticity in response to one factor (e.g. light) can be affected by another factor (e.g. water availability or herbivory pressure), so there are many ecological limits to phenotypic plasticity in plants (Valladares et al., 2007a). And examples relevant for understanding plant responses to climate change come from the acclimation to light. The light environment of the plant has been shown to affect the sensitivity of respiration to short- and long-term changes in temperature under controlled conditions and also in the field where other abiotics factors also varied (Zaragoza-Castells et al., 2007a, b). Sun and shade leaves of the Mediterranean evergreen oak *Quercus ilex* were capable of approaching full acclimation to changes in the growth temperature. Seasonal changes in the thermal sensitivity (Q_{10}) of respiration were observed in this tree, with higher values in winter than in summer. However, while irradiance affected photosynthesis, it had no effect on the Q_{10} of leaf respiration although the latter rates were lower in shade-grown leaves than in their high-light grown counterparts (Zaragoza-Castells et al., 2007b). Nevertheless, *Quercus ilex* plants under shade showed respiratory thermal acclimation. Dynamics shifts of temperature response curves of respiration through the year were observed in the field under both sun and shade providing further evidence that these plants can acclimate (Zaragoza-Castells et al., 2007a). What this kind of studies reveal is that acclimation must be taken into account in order to establish accurate leaf gas exchange models in systems like these Mediterranean oak forests with very low carbon inputs.

In addition to ad-hoc plastic changes over the life of an individual, there is another type of change at the level the genes that is being caused by rapid climate change. Many studies are showing that phenotypic plasticity is not the only way species has to cope with climate change, and perhaps for some plants not even the most important one. It has been repeatedly reported over the past several decades that rapid climate change has led to heritable, genetic changes in plant populations

(Billington & Pelham, 1991; Etterson, 2004), so small plants with short life cycles and large population sizes will probably adapt to altered growing seasons and be able to persist (Franks et al., 2007).

Since plasticity of most tree species seems not able to compensate for the current rate of environmental change, the option would be to take advantage of the capacity of trees for microevolutionary change (Fig. 1) (Jackson, 2006). But even this microevolution and local adaptation is not enough to compensate for the rate of change so many species are either going extinct locally or moving upward or northward at rapid rates: 6.1 m and 6.1 km per decade respectively (Jump & Peñuelas, 2005; Parmesan, 2006). This poleward range shift has important implications (Parmesan, 2006). The response of species to changing environments is likely to be determined largely by population responses at range margins (Hampe & Petit, 2005). In contrast to the expanding edge, the low-latitude limit of species ranges remains understudied, and the critical importance of rear edge populations as long-term stores of species genetic diversity and foci of speciation is becoming to be more and more appreciated (Hampe, 2005; Jump et al., 2006). In fact low-latitude populations are often disproportionately important for the survival and evolution of forest species, and their ecological features, dynamics and conservation requirements differ from those of populations in other parts of the range.

5 Scaling Up to the Community: Species Interactions

All plants are killed by temperatures somewhere 46°C and 56°C, temperatures that are only found in nature near an unshaded soil in arid habitats (Körner, 2006). This fact drastically affect plant establishment in high irradiance environments and is the main reason for the initial requirement of some shading by the already established vegetation. This process by which some plants improves the conditions for other plants is named facilitation and it has been argued to be common among plants in stressful habitats (Bertness & Callaway, 1994). However, its generality in arid zones is far from absolute since plant-plant interactions dynamically switch from competition to facilitation and vice versa under still not well understood environmental conditions (Flores & Jurado, 2003; Maestre et al., 2006). Climate change is affecting the net balance of plant-plant interactions, shifting competition to facilitation and viceversa depending on local conditions (Maestre & Cortina, 2004; Maestre et al., 2005). Whether plants facilitate each other or compete against each other have profound implications in ecosystem functioning and it is a good prove of the importance of considering species interactions under climate change scenarios. Plant-plant interactions are known to play a key role in mediating the impacts of atmospheric nitrogen deposition, increased atmospheric carbon dioxide concentrations and climate change (Brooker, 2006).

Plant-plant interactions determine the regeneration of the forest. Many important tree species require other species to get established, specially at the southern or lower latitudinal or altitudinal range of their distribution. This is the case of

Scots Pine (*Pinus sylvestris*), which requires facilitation by shrubs to get established in dry areas of the Iberian Peninsula (Castro et al., 2004a). But facilitation translates into competition depending on the particular conditions of each year (Tielborger & Kadmon, 2000; Valladares & Pearcy, 2002; Valladares et al., 2007b) and there is no consensus on whether the shade cast by a potential nurse is always beneficial (Fig. 3, see discussions in Maestre et al., 2005, 2006). One way of solving the empirical discrepancies on the beneficial aspects of the shade is to consider the age of the protégé (i.e. the target plant). It is frequently the case that facilitation does take place in the very initial stages of plant germination and establishment, but as the protégé, there is a shift to competition (e.g. seedlings of *Pinus sylvestris* are initially facilitated by shrubs but then they compete with established trees; Castro et al., 2004a), which has been named the ontogenetic shift (Fig. 4, Miriti, 2006). I suggest that climate change will have a relatively higher impact on the initial stages of plant-plant interaction, making facilitation more transient and of a lower magnitude and, thus, decreasing the possibilities for forests to regenerate (Fig. 4). All this applies primarily to relatively dry ecosystems that are expected to become drier in the future. Interestingly, it has been recently shown in Mediterranean-type ecosystems that the ancestry of the lineage significantly determines the type of plant-plant interactions, with Tertiary species being facilitated by Quaternary species, the latter better adapted to the current levels of aridity (Valiente-Banuet et al., 2006).

But plant-plant interaction is just one case of the more general situation of multiple species interactions, which includes those involved in predation, herbivory, pollination and dispersal. In fact, the differential effect of climate change on each of these interacting species might have more profound impacts on ecosystem functioning than expected from single species studies (Peñuelas & Filella, 2001; Parmesan, 2006). Butterflies might move uphill to escape the increasing heat, but their host plant might not, so butterflies cannot feed and plants do not get pollinated (Wilson et al., 2005), and the same applies to many other plant-animal systems that climate change may uncouple (Fig. 1).

In a changing world, a complex network of interacting species is more likely to survive than a simple one with just a few interactions (Bascompte et al., 2006). And co-evolution becomes the key for understanding the chances of such a network to cope with environmental change. It is not only the trees that evolve, but all the co-occurring species and even the established interactions that evolve. Trees and their pathogens and herbivores are always changing, but antagonistic organisms usually have shorter generation times than the host trees, so they are more likely to outcompete trees in a long term evolutionary race. In support of this, it seems quite a general phenomenon that climate change is exacerbating the impact of pathogens with Dutch elm disease and chestnut blight as good forest examples (Harvell et al., 2002). Trees have, though, many ways to compensate for this asymmetric rate of evolution and, interestingly, their longevity leads to the formation of mutualisms, opening a totally new front against pathogens and herbivores (Petit & Hampe, 2006). The fact that we are only beginning to understand the complexity of coevolutionary biodiversity networks should not deter us from considering them in realistic analyses of the capacities of tree to cope with environmental change.

6 The Challenge of Modelling Distribution Responses to Climate Change

Climate change is a major threat for the maintenance of biological diversity worldwide, and modelling is a crucial tool for evaluating its overall impact and for accurate simulation of climatic scenarios and species potential ranges. The well-established relationships between temperature, precipitation and plant distribution included in global vegetation models allowed for direct predictions of the consequences of climate change on species distribution (Parmesan, 1996, 2006). And many of the predictions have been confirmed: the movement of forest species to higher elevations and latitudes as the climate to which they are adapted is displaced has been reported for numerous regions of the world (Camarero & Gutiérrez, 2004; Kullman, 2002; Lloyd & Fastie, 2003; Peñuelas & Boada, 2003; Sturm et al., 2001; Walther et al., 2002). Some general rules, such as the 6°C threshold temperature for plant development that establishes the treeline worldwide (Körner & Paulsen, 2004), seems to work well and can be easily incorporated in modelling exercises of plant distribution under new climate scenarios. However, there are many important uncertainties, such as the capacity and speed of acclimation and plasticity, which need, first, to be better understood and, second, explicitly accounted for in the models.

Mechanistic, and not only phenomenological, models are needed to advance our predictive capacity, but they must incorporate feedbacks due to the response of the organisms to the environmental change and not only the physiology underlying these responses. For instance, global warming is expected to increase evapotranspiration, causing soil moisture declines that can be more important than changes in precipitation in arid systems. But the models that predict this drying do not incorporate direct biotic responses to warming. Interestingly, the interactions between warming and the dominant biota in a grassland ecosystem produced the reverse effect on soil moisture, suggesting that declines in plant transpiration mediated by changes in phenology can offset direct increases in evaporative water losses under global warming (Zavaleta et al., 2003). The importance of phenotypic plasticity as a buffer against extinction has not been widely appreciated. In fact, the extent of species losses may have been overestimated in many simulations of distribution shifts induced by global change because the plasticity of species is not considered (Thuiller et al., 2005). Araujo and Guisan (2006) have revised the new challenges for modelling species distribution, suggesting the revision of the niche concept and the improvement of model parameterization among the five most important ones. However, they did not explicitly mention mechanisms, although they were somehow included under the niche concept, which is in turn highly contested (Gravel et al., 2006). I argue that not only mechanisms must be incorporated into the modelling of the changes of species distributions but also that mechanisms must not be limited to ecophysiological ones: they should at least include the most essential ingredients of evolutionary biology (e.g. genetic diversity, rate of evolution, phenotypic plasticity) and community ecology (e.g. species interactions). The most immediate challenge, though, would be to learn how to incorporate them into the models.

Our knowledge on genetic diversity, phenotypic plasticity, and ecophysiological performance of fragmented populations of trees in a changing climate is still very limited. Bioclimate envelope models can serve as a first approximation, but future management and conservation strategies require models that incorporate more detail and attain greater biological realism (Hampe, 2004; Hampe & Petit, 2005).

7 The Case of Mediterranean Forests

Most of our knowledge on forest ecology comes from temperate and tropical forests, but dryland forests such as those in Mediterranean-type regions have received much less attention. And this is not only a gap in scientific knowledge but also a serious limitation in our capacity to anticipate and mitigate the effect of climate change on forests because climate change is expected to affect these forests more dramatically than most temperate and tropical forests (Barker et al., 2007; Christensen et al., 2007). Mediterranean forests in a changing climate are exposed to at least two distinct features: (a) the unpredictability of the timing and intensity of drought, the most limiting factor, and, (b) the combination of several limiting abiotic factors, which involves functional trade-offs and imposes conflictive selective pressures. Given the magnitude of forecasted climatic trends, there are great concerns for the particularly rich biodiversity found in the region (Alcamo et al., 2007). These features, which can be shared to same extent by a number of forests worldwide, make Mediterranean forests a fitting study case of the challenges entailed by a changing climate. Besides, Mediterranean woody flora is represented by taxa originated under very different climatic conditions (Petit et al., 2005; Suc, 1984). Interestingly, the high biodiversity of the Mediterranean basin is due at least in part to the effects of previous climate changes since the region has in fact acted as a glacial refuge for many groups of species (Carrión et al., 2003). The present-day tree flora of the Mediterranean Basin is made up of both very resilient taxa that have already experienced many abrupt and intense climate changes in the past (Benito-Garzon et al., 2007; Petit et al., 2005), and of very vulnerable taxa that are climatically isolated and geographically restricted to places where local, more humid conditions allow them to survive (Hampe, 2005). The latter are interesting not only from a conservation point of view but also as early warning systems of climate change. Relict tree populations in the Mediterranean Basin represent an evolutionary heritage of disproportionate significance for the conservation of European plant biodiversity.

Though a considerably resilience in the face of abrupt climatic changes in the continent is a necessary common feature for relict species of tropical origin to persist (Petit et al., 2005), the functional attributes and the extent of phenotypic plasticity, local adaptation and genetic variability involved in their persistence are still poorly understood (Balaguer et al., 2001; Hampe, 2005; Hampe & Petit, 2005; Valladares et al., 2002). Mediterranean marginal populations of relict tree species usually concentrate in river belts, presumably because of down-slope habitat displacement from mountain slopes as the characteristic summer drought of these environments became more pronounced (Mejías et al., 2002). Despite the extremely low range filling in this drought-prone region, shift towards riparian habitats provides a likely explanation for the long term stability of peripheral populations. It is suggested that for these pre-Mediterranean species buffered range modification through habitat shift could be a widespread phenomenon, whose importance is likely to increase under the predicted decrease in precipitation (Pulido et al., 2007). For natural regeneration of many Mediterranean forest trees and shrubs, the shade cast by the established vegetation has been crucial. But climate change is making this shade too dry so the final balance of pros and cons of the Mediterranean shade might become more negative (Fig. 3), which may significantly change dynamics and long term stability of present day Mediterranean forests.

8 Concluding Remarks

Forests have been frequently exposed to important environmental changes over ample geological and historical periods of times, but the speed and the complex nature of the current global change impose a novel challenge that seems particularly hard to overcome. The intrinsically slow evolutionary rates of trees and the limits to their phenotypic plasticity imposed by complex environmental changes suggest a reduced capacity of forests to successfully cope with a rapid climate change coupled with many other simultaneous changes in the environment. Recent studies suggest that species with a long life cycle might not be able to cope with the rapid pace of climate change (Savolainen et al., 2004; Franks et al., 2007; for a review see Parmesan, 2006). However, our knowledge is clearly insufficient. We do not have a clear picture of the real drivers of climate and atmospheric changes and of all relevant climatic aspects that are changing beyond the global rise of temperatures. We are beginning to understand forest responses to changes in individual environmental factors, but many factors are simultaneously changing and they act in concert, and many species, which differ in their sensitivity and responsiveness to environmental change, co-occur and interact with each other leading to a complex network of responses. It is only after we fill some of these basic gaps that we will be able to understand forest trends in a changing world and to interpret their capacity to cope with the plethora of phenomena and processes involved in the notion of climate change. However, scientific understanding of forest sector mitigation options is sufficient for a prompt start and immediate implementation of the forestry mitigation activities implemented under the Kyoto Protocol (Nabuurs et al., 2007). As argued for the case of ecological restoration of degraded ecosystems (Valladares & Gianoli, 2007), we do not have to wait till all scientific uncertainties on patterns and processes of changing ecosystems are understood to mitigate and to adapt to the increasingly important ecological and socio-economical threats imposed by climate change.

Acknowledgements Thanks are due to the members of the Spanish thematic network GLOBIMED (www.globimed.net) for inspiring discussions on forests and global change. One anonymous referee helped to improve the text. Financial support was provided by the Spanish Ministry of Education and Science (ECOCLIM, CGL2007–66066-C04–02/BOS) and by the Programa de Actividades de I + D de la Comunidad de Madrid (Consejería de Educación) REMEDINAL-CM (S-0505/AMB/000335).

References

Alcamo, J., Moreno, J. M., Nováky, B., Bindi, M., Corobov, R., Devoy, R. J. N., et al. (2007). Europe: impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. v. d. Linden & C. E. Hanson (Eds.), *Climate change 2007* (pp. 541–580). Cambridge: Cambridge University Press.

Alonso, A., Valladares, F. (2007). International efforts on global change research. In E. Chuvieco (Ed.), *Earth observation of global change* (pp. 1–22). Dordrecht, The Netherlands: Springer.

Araujo, M. B., Guisan, A. (2006). Five (or so) challenges for species distribution modelling. *Journal of Biogeography, 33*, 1677–1688.

Atkin, O. K., Tjoelker, M. G. (2003). Thermal acclimation and the dynamic response of plant respiration to temperature. *Trends in Plant Science, 8*, 343–361.

Atkin, O. K., Loveys, B. R., Atkinson, L. J., Pons, T. L. (2006). Phenotypic plasticity and growth temperature: understanding interspecific variability. *Journal of Experimental Botany, 57*(2), 267–281.

Balaguer, L., Martínez-Ferri, E., Valladares, F., Pérez-Corona, M. E., Baquedano, F. J., Castillo, F. J., et al. (2001). Population divergence in the plasticity of the response of *Quercus coccifera* to the light environment. *Functional Ecology, 15*, 124–135.

Ball, M. C., Hodges, V. S., Laughlin, G. P. (1991). Cold-induced photoinhibition limits regeneration of snow gum at tree-line. *Functional Ecology, 5*, 663–668.

Barker, T., Bashmakov, I., Bernstein, L., Bogner, J. E., Bosch, P. R., Dave, R., et al. (2007). Mitigation of climate change. Technical summary. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In B. Metz, O. R., Davidson, P. R., Bosch, R., Dave & L. A., Meyer (Eds.), *Climate change 2007* (pp. 1–103). Cambridge/New York: Cambridge University Press.

Bascompte, J., Jordano, P., Olesen, J. M. (2006). Asymmetric coevolutionary networks facilitate biodiversity maintenance. *Science, 312*, 431–433.

Beerling, D. J., Heath, J., Woodward, F. I., Mansfield, T. A. (1996). Drought-Co2 interactions in trees - observations and mechanisms. *New Phytologist, 134*(2), 235–242.

Benito-Garzon, M., Sanchez-de-Dios, R., Sainz-Ollero, H. (2007). Predictive modelling of tree species distributions on the Iberian Peninsula during the Last Glacial Maximum and Mid-Holocene. *Ecography, 30*, 120–134.

Bertness, M. D., Callaway, R. M. (1994). Positive interactions in communities. *Trends in Ecology and Evolution, 9*, 191–193.

Billington, H. L., Pelham, J. (1991). Genetic variation in the date of budburst in Scottish birch populations - implications for climate change. *Functional Ecology, 5*, 403–409.

Brooker, R. W. (2006). Plant-plant interactions and environmental change. *New Phytologist, 171*(2), 271–284.

Buchmann, N. (2002). Plant ecophysiology and forest response to global change. *Tree Physiology, 22*, 1177–1184.

Camarero, J. J., Gutiérrez, E. (2004). Pace and pattern of recent treeline dynamics: response of ecotones to climatic variability in the Spanish Pyrenees. *Climatic Change, 63*, 181–200.

Cannell, M. G. R., Thornley, J. H. (2000). Modelling the components of plant respiration: some guiding principles. *Annals of Botany, 85*, 45–54.

Cannell, M. G. R., Grace, J., Booth, A. (1989). Possible impacts of climatic warming on trees and forests in the united kingdom: a review. *Forestry, 62*, 337–364.

Carrión, J. S., Yll, E. I., Walker, M. J., Legaz, A., Chaín, C., López, A. (2003). Glacial refugia of temperate, Mediterranean and Ibero-North African flora in southeastern Spain: new evidence from cave pollen at two Neanderthal man sites. *Global Ecology and Biogeography, 12*, 119–129.

Castro, J., Zamora, R., Hodar, J. A., Gomez, J. M. (2004a). Seedling establishment of a boreal tree species (Pinus sylvestris) at its southernmost distribution limit: consequences of being in a marginal Mediterranean habitat. *Journal of Ecology, 92*(2), 266–277.

Castro, M., Martín-Vide, S., Alonso, S. (2004b). El clima de España: pasado, presente y escenarios de clima para el siglo XXI. In J. M. Moreno (Ed.), *Evaluación de los impactos del cambio climático en España* (pp. 3–64). Madrid: Ministerio de Medio Ambiente.

Christensen, J. H., Hewitson, B., Busuioc, A., Chen, A., Gao, X., Held, I., et al. (2007). Regional climate projections. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor & H. L. Miller (Eds.), *Climate change 2007: The physical science basis* (pp. 847–943). Cambridge/New York: Cambridge University Press.

Diamond, J. (2005). *Collapse: How societies choose to fail or succeed*. New York: Viking.

Dullinger, S., Dirnbock, T., Grabherr, G. (2004). Modelling climate change-driven treeline shifts: relative effects of temperature increase, dispersal and invasibility. *Journal of Ecology, 92*(2), 241–252.

Etterson, J. R. (2004). Evolutionary potential of Chamaecrista fasciculata in relation to climate change. II. Genetic architecture of three populations reciprocally planted along an environmental gradient in the great plains. *Evolution, 58*(7), 1459–1471.

Flores, J. L. F., Jurado, E. (2003). Are nurse-protégé interactions more common among plants from arid environments? *Journal of Vegetation Science, 14*, 911–916.

Franks, S. J., Sim, S., Weis, A. E. (2007). Rapid evolution of flowering time by an annual plant in response to a climate fluctuation. *Proceedings of the National Academy of Sciences USA, 104*, 1278–1282.

Gedney, N., Cox, P. M., Betts, R. A., Boucher, O., Huntingford, C., Stott, P. A. (2006). Detection of a direct carbon dioxide effect in continental river runoff records. *Nature, 439*, 835–838.

Gómez-Aparicio, L., Valladares, F., Zamora, R. (2006). Differential light responses of Mediterranean tree saplings: linking ecophysiology with regeneration niche in four co-occurring species. *Tree Physiology, 26*, 947–958.

Grace, J. (2004). Understanding and managing the global carbon cycle. *Journal of Ecology, 92*(2), 189–202.

Grace, J., Zhang, R. (2006). Predicting the effect of climate change on global plant productivity and the carbon cycle. In J. I. L. Morison & M. D. Morecroft (Eds.), *Plant growth and climate change* (pp. 187–207). Kundli, India: Blackwell.

Gravel, D., Canham, C. D., Beaudet, M., Messier, C. (2006). Reconciling niche and neutrality: the continuum hypothesis. *Ecology Letters, 9*, 399–409.

Greenland, D., Goodin, D. G., Smith, R. C. (Eds.) (2003). *Climate variability and ecosystem response in long-term ecological research sites*. New York: Oxford University Press.

Gutschick, V. P., BassiriRad, H. (2003). Extreme events as shaping physiology, ecology, and evolution of plants: toward a unified definition and evaluation of their consequences. *New Phytologist, 160*(1), 21–42.

Hampe, A. (2004). Bioclimate envelope models: what they detect and what they hide. *Global Ecology and Biogeography Letters, 13*, 469–476.

Hampe, A. (2005). Fecundity limits in *Frangula alnus* (Rhamnaceae) relict populations at the species' southern range margin. *Oecologia, 143*, 377–386.

Hampe, A., Petit, R. J. (2005). Conserving biodiversity under climate change: the rear edge matters. *Ecology Letters, 8*, 461–467.

Harvell, C. D., Mitchell, C. E., Ward, J. R., Altizer, S., Dobson, A. P., Ostfeld, R. S., et al. (2002). Climate warming and disease risks for terrestrial and marine biota. *Science, 296*, 2158–2162.

Howe, G. T., Aitken, S. N., Neale, D. B., Jermstad, K. D., Wheeler, N. C., Chen, T. H. (2003). From genotype to phenotype: unraveling the complexities of cold adaptation in forest trees. *Canadian Journal of Forest Research, 81*, 1247–1266.

Inouye, D. W. (2000). The ecological and evolutionary significance of frost in the context of climate change. *Ecology Letters, 3*, 457–463.

Jackson, S. T. (2006). Forest genetics in space and time. *New Phytologist, 171*(1), 1–3.

Joffre, R., Rambal, S., Winkel, T. (2001). Respuestas de las plantas mediterráneas a la limitación de agua: desde la hoja hasta el dosel. In R. Zamora & F. I. Pugnaire (Eds.), *Aspectos funcionales de los ecosistemas mediterráneos* (pp. 37–85). Granada: CSIC-AEET.

Jump, A. S., Peñuelas, J. (2005). Running to stand still: adaptation and the response of plants to rapid climate change. *Ecology Letters, 8*, 1010–1020.

Jump, A. S., Peñuelas, J. (2006). Genetic effects of chronic habitat fragmentation in a wind-pollinated tree. *Proceedings of the National Academy of Sciences USA, 103*, 8096–8100.

Jump, A. S., Hunt, J. M., Peñuelas, J. (2006). Rapid climate change-related growth decline at the southern range edge of *Fagus sylvatica. Global Change Biology, 12*, 1–12.

Körner, C. (2003a). Carbon limitation in trees. *Journal of Ecology, 91*, 4–17.

Körner, C. (2003b). Limitation and stress - always or never? *Journal of Vegetation Science, 14*, 141–143.

Körner, C. (2006). Significance of temperature in plant life. In J. I. L. Morison & M. D. Morecroft (Eds.), *Plant growth and climate change* (pp. 48–69). Kundli, India: Blackwell.

Körner, C., Paulsen, J. (2004). A world-wide study of high altitude treeline temperatures. *Journal of Biogeography, 31*, 713–732.

Kullman, L. (2002). Rapid recent range-margin rise of tree and shrub species in the Swedish Scandes. *Journal of Ecology, 90*, 68–77.

Lambers, H., Chapin III, F. S., Pons, T. L. (1998). *Plant physiological ecology*. York, PA: Springer.

Larcher, W. (1995). *Physiological plant ecology: Ecophysiology and stress physiology of functional groups*. Berlin/Heidelberg: Springer.

Lloret, F., Peñuelas, J., Estiarte, M. (2004). Experimental evidence of reduced diversity of seedlings due to climate modification in a Mediterranean-type community. *Global Change Biology, 10*(2), 248–258.

Lloyd, A. H., Fastie, C. L. (2003). Recent changes in treeline forest distribution and structure in interior Alaska. *Ecoscience, 10*, 176–185.

Maestre, F. T., Cortina, J. (2004). Do positive interactions increase with abiotic stress? A test from a semi-arid steppe. *Proceedings of the Royal Society of London Series B-Biological Sciences, 271*, S331–S333.

Maestre, F. T., Valladares, F., Reynolds, J. F. (2005). Is the change of plant-plant interactions with abiotic stress predictable? A meta-analysis of field results in arid environments. *Journal of Ecology, 93*, 748–757.

Maestre, F. T., Valladares, F., Reynolds, J. F. (2006). The stress-gradient hypothesis does not fit all relationships between plant–plant interactions and abiotic stress: further insights from arid environments. *Journal of Ecology, 94*, 17–22.

Martínez-Alonso, C., Valladares, F., Camarero, J. J., López Arias, M., Serrano, M., Rodríguez, J. A. (2007). The uncoupling of secondary growth, cone and litter production by intradecadal climatic variability in a Mediterranean Scots Pine Forest. *Forest Ecology and Management,* vol 253:19–29.

Matesanz, S., Brooker, R. W., Valladares, F., Klotz, S. (2008). Effects of long-term climate change on marginal steppic vegetation: from the community to the species. *Journal of Vegetation Science*.

Meehl, G. A., Stocker, T. F., Collins, W. D., Friedlingstein, P., Gaye, A. T., Gregory, J. M., et al. (2007). Global climate projections. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor & H. L. Miller (Eds.), *Climate change 2007: The physical science basis* (pp. 747–847). Cambridge/New York: Cambridge University Press.

Mejías, J. A., Arroyo, J., Ojeda, F. (2002). Reproductive ecology of Rhododendron ponticum (Ericaceae) in relict Mediterranean populations. *Botanical Journal of the Linnean Society, 140*, 297–311.

Menzel, A., Sparks, T. H., Estrella, N., Koch, E., Aasa, A., Ahas, R., et al. (2006). European phenological response to climate change matches the warming pattern. *Global Change Biology, 12*, 1969–1976.

Miriti, M. N. (2006). Ontogenetic shift from facilitation to competition in a desert shrub. *Journal of Ecology, 94*, 973–979.

Mittler, R. (2006). Abiotic stress, the field environment and stress combination. *Trends in Plant Science, 11*, 15–19.

Mooney, H. A., Winner, W. E., Pell, E. J. (Eds.) (1991). *Response of plants to multiple stresses*. San Diego, CA: Academic.

Morales, P., Sykes, M. T., Prentice, I. C., Smith, P., Smith, B., Bugmann, H., et al. (2005). Comparing and evaluating process-based ecosystem model predictions of carbon and water fluxes in major European forest biomes. *Global Change Biology, 11*, 2211–2233.

Nabuurs, G. J., Masera, O., Andrasko, K., Benitez-Ponce, P., Boer, R., Dutschke, M., et al. (2007). Forests. Impacts, adaptation and vulnerability. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. v. d. Linden & C. E. Hanson (Eds.), *Climate change 2007* (pp. 1–73). Cambridge: Cambridge University Press.

Niinemets, Ü. (2006). The controversy over traits conferring shade-tolerance in trees: ontogenetic changes revisited. *Journal of Ecology, 94*, 464–470.

Niinemets, U., Valladares, F. (2006). Tolerance to shade, drought and waterlogging of temperate, Northern hemisphere trees and shrubs. *Ecological Monographs, 76*, 521–547.

Ogaya, R., Peñuelas, J. (2003). Comparative field study of *Quercus ilex* and *Phillyrea latifolia*: photosynthetic response to experimental drought conditions. *Environmental and Experimental Botany, 50*, 137–148.

Parmesan, C. (1996). Climate and species range. *Nature, 382*, 765–766.

Parmesan, C. (2006). Ecological and evolutionary responses to recent climate change. *Annual Review of Ecology, Evolution, and Systematics, 37*, 637–669.

Peñuelas, J., Boada, M. (2003). A global change-induced biome shift in the Montseny mountains (NE Spain). *Global Change Biology, 9*, 131–140.

Peñuelas, J., Filella, I. (2001). Phenology: responses to a warming world. *Science, 294*, 793–795.

Petit, R. J., Hampe, A. (2006). Some evolutionary consequences of being a tree. *Annual Review of Ecology, Evolution and Systematics, 37*, 187–214.

Petit, R. J., Hampe, A., Cheddadi, R. (2005). Climate changes and tree phylogeography in the Mediterranean. *Taxon, 54*(4), 877–885.

Piersma, T., Drent, J. (2003). Phenotypic flexibility and the evolution of organismal design. *Trends in Ecology and Evolution, 18*, 228–233.

Pinker, R. T., Zhang, B., Dutton, E. G. (2005). Do satellites detect trends in surface solar radiation? *Science, 308*, 850–853.

Pulido, F., Valladares, F., Calleja, J. A., Moreno, G., González-Bornay, G. (2008). Tertiary relict trees under Mediterranean climate: abiotic constraints on persistence of *Prunus lusitanica* at the eroding edge of its range. *Journal of Biogeography* doi: 10.1111/j.1365-2699.2008.01898.x.

Rambal, S., Joffre, R., Ourcival, J. M., Cavender-Bares, J., Rocheteau, A. (2004). The growth respiration component in eddy CO2 flux from a *Quercus ilex* Mediterranean forest. *Global Change Biology, 10*(9), 1460–1469.

Rehfeldt, G., Wykoff, W. R., Ying, C. C. (2001). Physiological plasticity, evolution, and impacts of a changing climate on *Pinus contorta. Climatic Change, 50*, 355–376.

Reichstein, M., Tenhunen, J. D., Roupsard, O., Ourcival, J. M., Rambal, S., Miglietta, F., et al. (2002). Severe drought effects on ecosystem CO2 and H2O fluxes at three Mediterranean evergreen sites: revision of current hypotheses? *Global Change Biology, 8*(10), 999–1017.

Richardson, A. D., Bailey, A. S., Denny, E. G., Martin, C. W., O'Keefe, J. (2006). Phenology of a northern hardwood forest canopy. *Global Change Biology, 12*(7), 1174–1188.

Roderick, M. L., Farquhar, G. D. (2005). Changes in New Zealand pan evaporation since the 1970s. *International Journal of Climatology, 25*, 2031–2039.

Roderick, M. L., Farquhar, G. D., Berry, S. L., Noble, I. R. (2001). On the direct effect of clouds and atmospheric particles on the productivity and structure of vegetation. *Oecologia, 129*, 21–30.

Sánchez-Gómez, D., Valladares, F., Zavala, M. A. (2006a). Performance of seedlings of Mediterranean woody species under experimental gradients of irradiance and water availability: trade-offs and evidence for niche differentiation. *New Phytologist, 170*, 795–806.

Sánchez-Gómez, D., Zavala, M. A., Valladares, F. (2006b). Seedling survival responses to irradiance are differentially influenced by low-water availability in four tree species of the Iberian cool temperate–Mediterranean ecotone. *Acta Oecologica, 30*, 322–332.

Sanz-Perez, V., Castro-Diez, P., Valladares, F. (2008). Differential and interactive effects of temperature and photoperiod on budburst of two co-occurring Mediterranean oaks. *Tree Physiology* (in press).

Savolainen, O., Bokma, F., Garcia-Gil, R., Komulainen, P., Repo, T. (2004). Genetic variation in cessation of growth and frost hardiness and consequences for adaptation of *Pinus sylvestris* to climatic changes. *Forest Ecology and Management, 197*, 79–89.

Saxe, H., Cannell, M. G. R., Johnsen, O., Ryan, M. G., Vourlitis, G. (2001). Tree and forest functioning in response to global warming. *New Phytologist, 149*, 369–400.

Stanhill, G., Cohen, S. (2001). Global dimming: a review of the evidence for a widespread and significant reduction in global radiation with discussion of its probable causes and possible agricultural consequences. *Agricultural and Forest Meterorology, 107*, 255–278.

Sturm, M., Racine, C., Tape, K. (2001). Climate change: increasing shrub abundance in the arctic. *Nature, 411*, 546–547.

Suc, J. P. (1984). Origin and evolution of the mediterranean vegetation and climate in Europe. *Nature, 307*, 429–432.

Taschler, D., Neuner, G. (2004). Summer frost resistance and freezing patterns measured in situ in leaves of major alpine plant growth forms in relation to their upper distribution boundary. *Plant, Cell and Environment*, 737–746.

Thuiller, W., Lavorel, S., Araujo, M. B., Sykes, M. T., Prentice, I. C. (2005). Climate change threats to plant diversity in Europe. *PNAS, 102*, 8245–8250.

Tielborger, K., Kadmon, R. (2000). Temporal environmental variation tips the balance between facilitation and interference in desert plants. *Ecology, 81*(6), 1544–1553.

Valiente-Banuet, A., Vital-Rumebe, A., Verdu, M., Callaway, R. M. (2006). Modern Quaternary plant lineages promote diversity through facilitation of ancient Tertiary lineages. *Proceedings of the National Academy of Sciences USA, 103*, 16812–16817.

Valladares, F. (2003). Light heterogeneity and plants: from ecophysiology to species coexistence and biodiversity. In K. Esser, U. Lüttge, W. Beyschlag & F. Hellwig (Eds.), *Progress in Botany* (Vol. 64, pp. 439–471). Heidelberg: Springer.

Valladares, F. (2004a). Global change and radiation in Mediterranean forest ecosystems: a meeting point for ecology and management. In M. Arianoutsou & V. Papanastasis (Eds.), *Ecology, conservation and sustainable management of mediterranean type ecosystems of the World* (pp. 1–4). Rotterdam: Millpress.

Valladares, F. (Ed.) (2004b). *Ecología del bosque mediterraneo en un mundo cambiante*. Madrid: Organismo Autónomo de Parques Nacionales, Ministerio de Medio Ambiente.

Valladares, F., Gianoli, E. (2007). How much ecology do we need to know to restore Mediterranean ecosystems? *Restoration Ecology, 15*, 363–368.

Valladares, F., Niinemets, Ü. (2008). Shade tolerance, a key plant trait of complex nature and consequences. *Annual Review of Ecology, Evolution and Systematics, 39*, 343–366.

Valladares, F., Pearcy, R. W. (1997). Interactions between water stress, sun-shade acclimation, heat tolerance and photoinhibition in the sclerophyll *Heteromeles arbutifolia. Plant, Cell and Environment, 20*, 25–36.

Valladares, F., Pearcy, R. W. (2002). Drought can be more critical in the shade than in the sun: a field study of carbon gain and photoinhibition in a Californian shrub during a dry El Niño year. *Plant Cell and Environment, 25*, 749–759.

Valladares, F., Sánchez-Gómez, D. (2006). Ecophysiological traits associated with drought in Mediterranean tree seedlings: individual responses versus interspecific trends in eleven species. *Plant Biology, 8*, 688–697.

Valladares, F., Balaguer, L., Martinez-Ferri, E., Perez-Corona, E., Manrique, E. (2002). Plasticity, instability and canalization: is the phenotypic variation in seedlings of sclerophyll oaks consistent with the environmental unpredictability of Mediterranean ecosystems? *New Phytologist, 156*, 457–467.

Valladares, F., Arrieta, S., Aranda, I., Lorenzo, D., Tena, D., Sánchez-Gómez, D., et al. (2005a). Shade tolerance, photoinhibition sensitivity and phenotypic plasticity of *Ilex aquifolium* in continental-Mediterranean sites. *Tree Physiology, 25*, 1041–1052.

Valladares, F., Dobarro, I., Sánchez-Gómez, D., Pearcy, R. W. (2005b). Photoinhibition and drought in Mediterranean woody saplings: scaling effects and interactions in sun and shade phenotypes. *Journal of Experimental Botany, 56*, 483–494.

Valladares, F., Sanchez, D., Zavala, M. A. (2006). Quantitative estimation of phenotypic plasticity: bridging the gap between the evolutionary concept and its ecological applications. *Journal of Ecology, 94*, 1103–1116.

Valladares, F., Gianoli, E., Gómez, J. M. (2007a). Ecological limits to plant phenotypic plasticity. Tansley review. *New Phytologist, 146,* 749–763.

Valladares, F., Zaragoza-Castells, J., Sánchez-Gómez, D., Matesanz, S., Alonso, B., Portsmuth, A., et al. (2008). Is shade detrimental for Mediterranean shrubs experiencing periods of extreme drought and late-winter frosts? *New Phytologist Journal*: *Annals of Botony*.

Walther, G. R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T. J. C., et al. (2002). Ecological responses to recent climate change. *Nature, 426*, 389–395.

Wild, M., Gilgen, H., Roesch, A., Ohmura, A., Long, C. N., Dutton, E. G., et al. (2005). From dimming to brightening: decadal changes in solar radiation at Earth's surface. *Science, 308*, 847–850.

Wilson, R., Gutierrez, D., Gutierrez, J., Martínez, D., Agudo, R., Monserrat, V. J. (2005). Changes to the elevational limits and extent of species ranges associated with climate change. *Ecology Letters, 8*, 1138–1146.

Woodward, F. I. (1987). Stomatal numbers are sensitive to increases in CO2 from pre-industrial levels. *Nature, 327*, 617–618.

Zaragoza-Castells, J., Sánchez-Gómez, D., Hartley, I. P., Matesanz, S., Valladares, F., Lloyd, J., et al. (2007a). Climate-dependent variations in leaf respiration in a dry-land, low productivity Mediterranean forest: the importance of thermal acclimation in both high-light and shaded habitats. *Functional Ecology, 22*(1), 172–184.

Zaragoza-Castells, J., Sánchez-Gómez, D., Valladares, F., Hurry, V., Atkin, O. K. (2007b). Does growth irradiance affect temperature-dependence and thermal acclimation of leaf respiration? Insights from a Mediterranean tree with long-lived leaves. *Plant, Cell and Environment, 30*, 820–833.

Zavaleta, E. S., Thomas, B. D., Chiariello, N. R., Asner, G. P., Shaw, M. R., Field, C. B. (2003). Plants reverse warming effect on ecosystem water balance. *Proceedings of the National Academy of Sciences USA, 100*, 9892–9893.

Ziska, L. H., Bunce, J. A. (2006). Plant responses to rising atmospheric carbon dioxide. In J. I. L. Morison & M. D. Morecroft (Eds.), *Plant growth and climate change* (pp. 17–47). Kundli, India: Blackwell.

Greenhouse-gas Emissions from Temperate Mountain Forests

R. Jandl[1]*, G. Wieser[2], F. Hagedorn[3], and A. Schindlbacher[1]

1 Introduction

Mountain forests are home to a stunning diversity of animals and plants. They contribute if not define the scenic beauty of a landscape in a harsh environment. Forests fulfil many environmental services. An example is the regulation of hydrologic processes that tames otherwise dangerous torrents and converts them into sources of drinking water and enables to transform the energy within hydrologic power stations. An only recently emerging issue is whether or not mountain forests can contribute to the sequestration of greenhouse gases (GHGs).

Mountain regions are often covered by forests because site properties, environmental conditions and limited land accessibility impose difficulties to the agricultural land use. During the last 50 years, mountain forests probably underwent the most accentuated and significant change in their history. For centuries, timber extraction was the dominant aim of forestry and patches of land were cleared for pastures in order to seasonally expand the agricultural land from valleys into higher elevation. The different products of the management of mountain forests served a local market. The historical economical context of forestry between timber production and protection of infrastructure and settlements has changed since tourism exerts a steadily growing demand on land. Forestry as a part of the primary sector of economy is losing on relevance. The local economy no longer depends on locally produced timber. The high costs of timber production (silvicultural interventions plus cost of harvesting) in mountain regions further marginalize forestry. The traditional intensity of forestry in mountains is no longer economically viable and intermittent forms of forestry, i.e. brief episodes of active forest management and long intervals without interventions, are a realistic scenario (Broggi, 2002).

---

[1] Federal Research and Training Centre for Forests, Natural Hazards, and Landscape (BFW), A-1131 Vienna, Austria. *robert.jandl@bfw.gv.at

[2] Federal Research and Training Centre for Forests, Natural Hazards, and Landscape (BFW), A-6020 Innsbruck, Austria

[3] Swiss Federal Institute for Forest, Snow and Landscape Research, 8903 Birmensdorf, Switzerland

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

Under the current socio-economic conditions, the value of mountain forests for the storage of GHGs is of high relevance. Climatic change re-defines the environmental conditions and mountain ecosystems are listed among the most vulnerable types of ecosystems (IPCC, 2001, 2007). Evidence for climate-change effects on mountain plants is long known (Grabherr & Pauli, 1994; Dullinger et al., 2004 and many others). Climatic change is a phenomenon with many facets. While there is a general agreement about rising temperatures as a consequence of anthropogenic combustion processes, scenarios about future precipitation patterns and the frequency of ecosystem disturbances are less certain (Matulla et al., 2003). Predictions on the response of mountain forests to climatic change are uncertain. The timberline will certainly continue to rise and the pattern of snow coverage will change (Kullmann, 2002). Such changes have happened before (Nicolussi et al., 2005) and had dramatic consequences for the local settlements, but were not more than marginal notes outside the region. On a first glance the response of the entire ecosystem is very simple: extension of a dense forest cover increases aboveground biomass and leads to forest soils that are rich in soil organic matter. However, several factors act independent of each other and both the biogeochemical cycles of several elements and the disturbance regime of the ecosystem are affected. The joint effect of these factors on the development of the ecosystem over decades remains speculative (Loya & Grogan, 2004).

In this chapter we establish that mountain forest ecosystems in the temperate region contain large C and N pools that are quite active and can be mobilized under certain conditions. We provide information on the peculiarities of mountain forests with respect to GHG dynamics and give an overview of the results of field experiments and modeling exercises indicating that forests can be both sinks and sources of GHGs.

The GHG that receives most attention is CO_2. The biogeochemical C cycle includes the fixation of CO_2 via photosynthesis, its incorporation into biomass and soil organic C, and its release by autotrophic and heterotrophic respiration and in smaller amounts by leaching of dissolved organic C (DOC). Annually, large quantities are exchanged between the atmosphere and terrestrial ecosystems by photosynthesis and respiration.

In an ecosystem at equilibrium, these fluxes are in balance. The system neither loses nor gains C. Climate change and improved forms of forest management set new conditions and the equilibrium level of C is thought to rise currently (Janssens et al., 2003; Vetter et al., 2005), and thus, mountain forests act as a C sink. In Switzerland, modelling the effects of increasing litter inputs to soils during the last decades suggests that soils contribute to 30% to the C sequestration in mountain forest ecosystems (Perruchoud et al., 1999).

Other GHGs that can be released in high quantities from the soils of mountains forest ecosystems are nitrogen oxides such as N_2O and NO_x ($NO + NO_2$). The amount of nitrogen oxides released from mountain forest soils is marginal, but the radiative forcing is high. N_2O is a direct greenhouse gas with an atmospheric lifetime of 120 years. Its 100-year global warming potential is about 300 times as high as that of carbon dioxide. In contrast, NO_x are not GHGs themselves but their indirect positive radiative forcing through catalysing the production of tropospheric O_3, the third most important GHG, is relevant (IPCC, 2007). Therefore, even low emissions of nitrogen oxides matter. Apparently, there is a function of forests with a high relevance for the society due to its contribution to the terrestrial C sink. It is important to understand under which environmental and economic conditions this 'environmental service' of forests is maintained. Other GHGs such as CH_4 are of minor relevance in mountain forests and are not treated in this chapter.

In this text we evaluate the effects of climatic change on GHG emissions from mountain forests. Surprisingly few data about GHG storage in mountain forest ecosystems exists, but a plethora of experiments have created an agreement that forest ecosystems play an important role. Estimations for global or even regional GHG fluxes are highly uncertain and fundamental data are still missing. Alone the estimate for the annual global rate of C respiration from soils has been updated within 10 years from 75 to 50 Petagrams (Schlesinger & Andrews, 2000; Field et al., 2004). Here, we will focus on the GHG fluxes and their controls in mountain forest ecosystems.

In comparison to low elevation forests, mountain forests generally have

a. Harsher climates with lower air temperature, shorter growing seasons and higher precipitation, implying that C-storage and GHG budget will response sensitive to the expected warming. Snow cover is an important co-determinant for winter temperatures.
b. Soils are less developed, indicating that less soil organic carbon is stabilized in mineral soils, but more C is stored in labile forms that are susceptible to losses via respiration.
c. Mountain forests are less intensively managed, and natural disturbances such as landslides and avalanches are more important than man-made disturbances.

This description is not exhaustive and applies for ecosystems in the temperate region (Tranquillini, 1979; Körner, 2003). Current land-use changes are larger than in the lowlands, implying that these changes will play a major role in GHG budgets.

Mountain forest ecosystems store an enormous amount of C and N in their biomass and in soils. These elements can be both sequestered and released. During the last decades, the productivity of mountain forests has increased (Spiecker et al., 1996; Paulsen et al., 2000) as a result of several factors such as nitrogen deposition, a longer growing season, CO_2 fertilization, and adapted forms of forest management. In addition, the current changes in land-use change are drastic in Mid European mountain forest. For instance, in Switzerland the forested area increases by 1% per year in altitudes above 1,400 m as compared to 0.2% in the lower altitudes. Consequently, both growth and area of mountain forests are currently increasing thereby altering the ecosystem GHG budget.

Information on mountain forests can be gained from a large number of experiments in the boreal region. The altitudinal gradient resembles in many aspects the northward gradient and is especially a valid thermal gradient. Accordingly, among the similarities are low productivity and short growing seasons. However, geomorphology is an important factor as well. Soils in boreal forests are more often limited by excess soil moisture and lack of temperature. In the Alps, steep slopes, and high content of coarse soil constituents (rocks) are favourable for drainage, and frequent precipitation events ensure sufficient soil moisture. Under such conditions temperature is the major limiting factor.

2 Carbon and Nitrogen Pools in Mountain Forests

Mountain forests soils have higher stone contents and are shallower than soils in low elevation. On silicatic bedrock the dominant soil formations are Cambisols and Spodosols in different stages of their development, on calcareous limestone Rendzic Leptosols are commonly found (FAO soil classification). Figure 1 shows that high elevation soils contain similar amounts or even more C and N than soils in low elevation. Soil texture is an important soil property. Figure 2 shows that most soils in high elevation are, irrespective of the bedrock material, poor in clay content. The soil organic matter plays therefore an important role for the sorption of nutrients. Any climate-driven decline in soil organic matter will have consequences for site productivity. The aboveground forest biomass declines with altitude, as forests tend to be smaller and the canopy coverage is lower. The proportion of forest land is low in low elevation and increases up to timberline (1,500–2,200 m in Austria) (Fig. 1, right part).

Figure 3 shows that the CO_2 emission from soils responds to increased temperatures at all elevations. Similar results apply for the release of NO and N_2O. No clear elevational gradient can be seen, indicating that soil C and N from high elevation

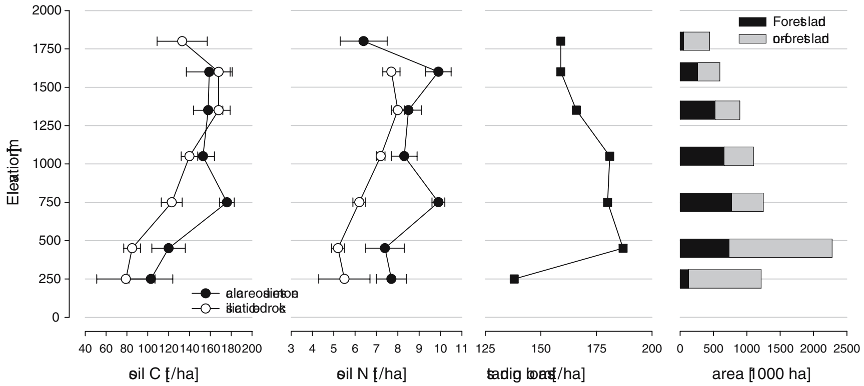


Fig. 1 Soil carbon and nitrogen pool, standing biomass (stems, branches, needles of conifers), forest area and non-forest area of Austria. (Data sources: Soil data: Austrian Forest Soil Observation System; Biomass and land use: Austrian National Forest Inventory)

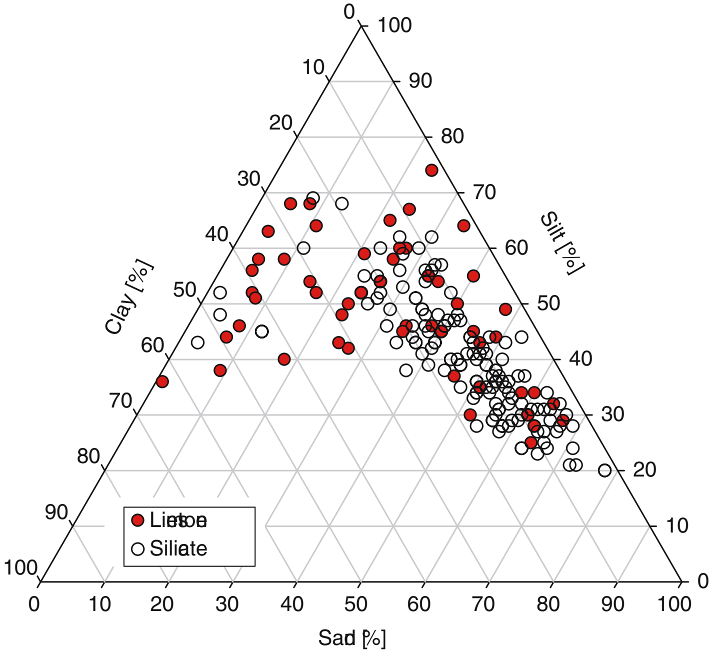


Fig. 2 Size class distribution of Austrian forest soils from elevations above 1,000 m. (Data source: Austrian Forest Soil Survey)

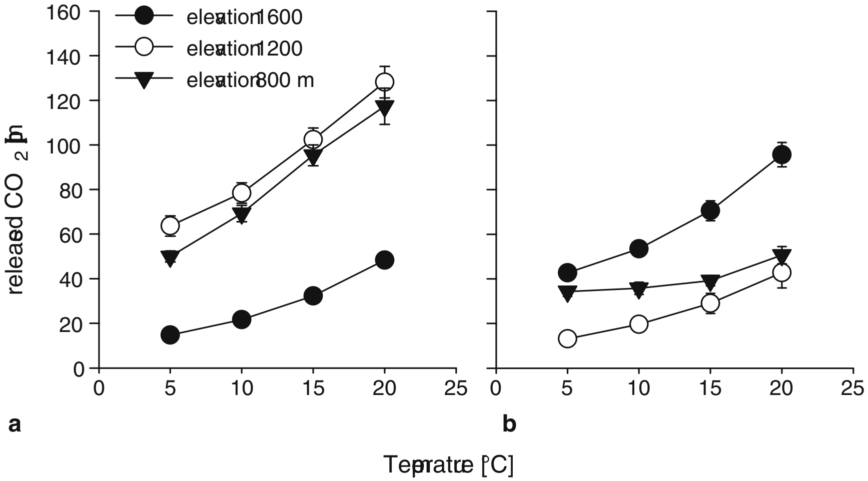


Fig. 3 Release of CO_2 from (a) the forest floor material and (b) the mineral soil collected at different elevations. The samples were incubated in the laboratory at 4°C, 8°C, 16°C, and 20°C

are not protected to a higher degree by some stabilization mechanism (de Gonzalo Aranoa, 2006). We expect that increased temperatures will bring mountain soils to a new equilibrium status in their C and N pools. Especially due to global warming, for some period of time these soils will be strong sources of GHGs. A potential remedy is the application adapted forest management strategies that are optimized towards the increase or at least maintenance of the pool of soil organic matter (Jandl et al., 2006).

3 Net Ecosystem CO_2 and H_2O Exchange in High Elevation Forests

Net ecosystem CO_2 exchange is the difference between gross primary production (GPP) and the sum of autotrophic and heterotrophic respiration (= total ecosystem respiration; ER). On an annual basis, high elevation forest ecosystems generally show a small net gain in the carbon exchange with the atmosphere (Zeller & Nikolov, 2000; Carey et al., 2001; Monson et al., 2002), and thus representing net ecosystem production (NEP) or carbon sequestration.

A full seasonal carbon balance for a 95-year-old Pinus cembra forest (stand density 1,038 trees per hectare, basal area 9.5 m^2 ha^{-1}, projected leaf area of 3.9 m^2 m^{-2}) at timberline in the Austrian Alps is shown in Fig. 4. The daily carbon (C) balance of this forest ecosystem was negative from November up to mid April (≈5 months) and it took a further 1.5 months to compensate the total of 82 g C m^{-2} surface area lost by ER during the period of dormancy.

The annual GPP of this open *Pinus cembra* forest was 1,610 g C m^{-2} (Table 1), which is within the range of 831 to 1,690 g C m^{-2} $year^{-1}$ estimated for other high altitude and high latitude coniferous forests (Valentini et al., 2000; Falge et al., 2002; Zha et al., 2004). Total annual ER was 1,247 g C m^{-2} and was dominated by fluxes from aboveground components (Table 1). The low C efflux rates from belowground components of this ecosystem can be attributed to low root zone temperatures (Wieser & Bahn, 2004) as well as to a supra-optimal soil water availability as there is evidence that soil water potentials exceeding −0.05 MPa significantly depress soil surface CO_2 efflux in high elevation ecosystems (Siegwolf, 1987; Cernusca & Decker, 1989; Wieser, 2004a).

NEP of this pine forest was 363 g C m^{-2} $year^{-1}$ (Fig. 4) which is within the range reported for forest ecosystems at high elevation (58 to 500 g C m^{-2} $year^{-1}$; Zeller & Nikolov, 2000; Carey et al., 2001; Monson et al., 2002) and high latitude (−80 to 245 g C m^{-2} $year^{-1}$; Valentini et al., 2000) but significantly below the values estimated for latitudes lower than 51° N in Europe (470 to 600 g C m^{-2} $year^{-1}$; Valentini et al., 2000).

Throughout the growing season when C uptake was not limited by cold temperatures the temperature response curve of GPP is relatively wide as net photosynthesis operates at more than 90% of its maximum at a temperature range between 5°C and 20°C (Wieser, 2004b), and air temperature explains more than 60% of the seasonal

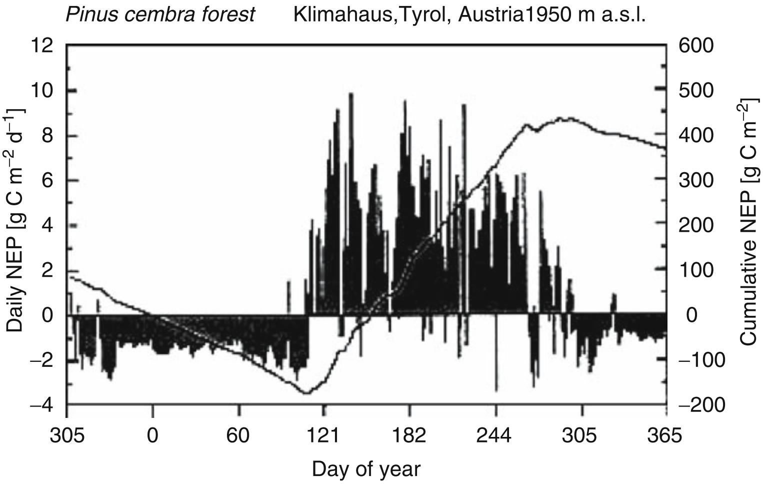


Fig. 4 Seasonal course of the daily NEP (grey bars) and the cumulative NEP (solid line) for a 95-year old Pinus cembra forest. Measurements were made at Mt. Patscherkofel (Klimahaus), Austria, 1,950 m a.s.l. (Modified after data from Wieser & Stöhr, 2005)

Table 1 Annual gross primary production (GPP), ecosystem respiration (ER), component respiration and net ecosystem production (NEP) (After Wieser & Stöhr, 2005)

Parameter	%
GPP	100 (= 1,610 g C m^{-2} year^{-1})
ER	77.5
Annual foliage respiration	17.1
Annual branch respiration	52.5
Annual stem respiration	2.8
Annual coarse root respiration	3.1
Annual small root respiration	1.4
Annual heterotrophic respiration	0.7
NEP	22.5

variation in GPP (Fig. 5). ER by contrast increased exponentially with temperature ($r^2 = 0.84$; Q_{10}: 2.45; base ER at 0°C: 2.11 g C m^{-2} day^{-1} Fig. 5). Thus, the relationship between GPP and ER (Fig. 5) strongly suggests that under current climatic conditions growing season temperatures and their effect on ER are the key factor limiting NEP in high elevation forests (cf. also Huxman et al., 2003). Low temperatures by contrast are not necessarily a burden to the overall C sequestration of high elevation forest ecosystems.

However, during the last decade the Earth's climate has warmed and global change models predict a further increase in mean air temperature of 1.4°C to 5.8°C in the next century (IPCC, 2001). On a regional scale this trend has also be documented at the Klimahaus Research Station at timberline on Mount Patscherkofel (Central Austrian Alps) where the mean annual air temperature during the last decade

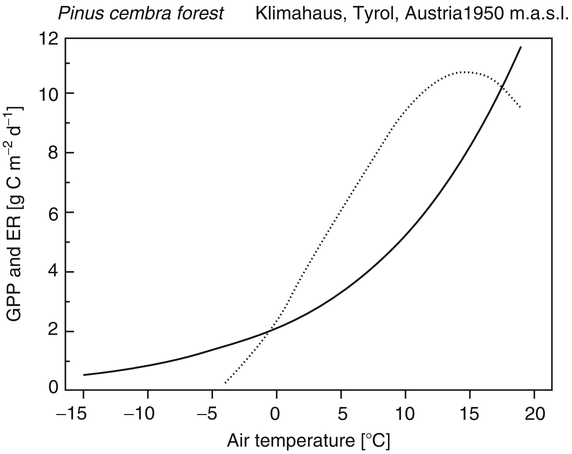


Fig. 5 The temperature dependence of gross primary production during the growing season (GPP; dotted line) and year round ecosystem respiration (ER solid line) in relation mean daily air temperature) in a 95-year old Pinus cembra forest. Measurements were made at Mt. Patscherkofel (Klimahaus), Austria, 1,950 m a.s.l. (Modified after data from Wieser & Stöhr, 2005)

(1995–2004) was at an average 1°C higher as compared to the 30 years before (Wieser & Tausz, 2006) and changes appear to be greatest during spring (+1.4°C) and summer (+1.6°C). Furthermore, at the timberline of the central Austrian Alps the growing season extended from 168 ± 12 days during the period 1972 to 1985 (Havranek, 1987) to 196 ± 23 days during the years 1994 to 2004 while growing season precipitation patterns have not changed significantly during the last 40 years (Wieser, 2004a). Similar trends apply for the entire Alps (Beniston, 2006).

Thus, carbon sequestration may change with predicted climate warming because respiration is relatively more sensitive to temperature than carbon gain through photosynthesis (Rayn, 1991). However, calculations based on the temperature responses shown in Fig. 5 suggest that an increase in mean annual air temperature of 1°C will increase GPP and ER by about 15% and 10%, respectively. Consequently, NEP of high elevation and high latitude forest ecosystems may benefit from future climate warming (Grace & Rayment, 2000; Strömgren & Linder, 2002; Zha et al., 2004)

4 Seasonal Dynamics

4.1 Carbon Dioxide

The C balance is the net product of two opposing processes. Photosynthesis fixes C, and respiration releases it from the ecosystem. The net ecosystem gain of C is rather low in mountain forests because of long, snowy winters and short vegetation periods in summer. Plant productivity is mainly limited by the length and temperatures of the growing season (Tranquillini, 1979; Körner, 1998). Snow depth in spring together with the length of the snow free time was found to be good indicators of tree growth. Once the snow pack is melted in spring, the rate of photosynthesis reaches high values, even when the air temperature is still low. At air Ts of 5°C, already 25% to 50% of the maximum rate of photosynthesis is reached and the optimum is already reached at approx 10°C (Körner, 2003; Huxman et al., 2003). Besides temperature and moisture, plant growth is also affected by the partial pressure of CO_2 and N deposition. Both factors enhance the growth rate, as long as sufficient soil moisture is available (Oren et al., 2001; Hungate et al., 2003; Wieser, 2004a,b). High altitude likely respond more sensitive to increasing atmospheric CO_2 than trees of the lowland since both gas exchange studies (Körner & Diemer, 1987) and global comparisons of stable carbon isotope data (Körner et al., 1991) show that the efficiency of CO_2 uptake at any given CO_2 partial pressure is higher in plants from high altitude. In a 5-year CO_2-enrichment study at the alpine treeline, Handa et al. (2005) found indeed that larch (*Larix decidua*) grew significantly better under elevated CO_2, while pine (*Pinus uncinata*) showed only a moderate initial response that disappeared thereafter. The growth response of both trees was significantly smaller than that of photosynthesis, indicating that the greater C-uptake was not only used for a higher growth rate.

Soil respiration, other than photosynthesis, is limited by the soil temperature and soil moisture. When soil moisture is abundant, as commonly observed in many mountain forests, the T limitation dominates. More often the soils are even too moist and respiration picks up when soil moisture decreases (Rodeghiero & Cescatti, 2005; Wieser, 2004a; Wieser & Stöhr, 2005).

4.2 Artificial Soil Warming

Experimental evidence for the future development of the CO_2 balance of mountain forests is given by a network of soil warming studies. They are expensive yet appropriate tools because the interaction between manifold processes in the C cycle is complex and difficult to predict. Ground truth is therefore required. Warming essentially affects all ecosystem processes. The system response to soil warming depends on stocks and initial turnover rates of labile carbon and nitrogen in the soil, relative size of the carbon pools of plants and soil, soil water regime, and the vegetation cover (Shaver et al., 2000). No simple answer is possible whether or not temperature can be seen as the single most important factor for the turnover of soil C, as substrate quality and provides an equally useful concept for the turnover of soil C (Davidson & Janssens, 2006). The overarching question is if soils under warmer conditions act as a continuous source of CO_2, or if and when a new equilibrium in the C pool establishes. In other words, the question is how much of the large soil C pool is available for decomposition by soil microbes or which part of the soil C pool is protected. A soil warming experiment in a low elevation forest in USA has shown a rapidly declining net CO_2 release. This effect became obvious after almost 10 years of experimental warming and underlines the importance of long-term experiments (Melillo et al., 2002).

It is required to evaluate the response of one compartment of the ecosystem, here the soil, together with others. Soil warming not only stimulates soil microbial processes, but also the growth of trees. An apparent rapid response, as suggested by Fig. 7 is therefore dampened and requires long-term experiments for a conclusive interpretation (Jarvis & Linder, 2000; Strömgren & Linder, 2002). An initial increase in autotrophic respiration is also expected to acclimate rapidly

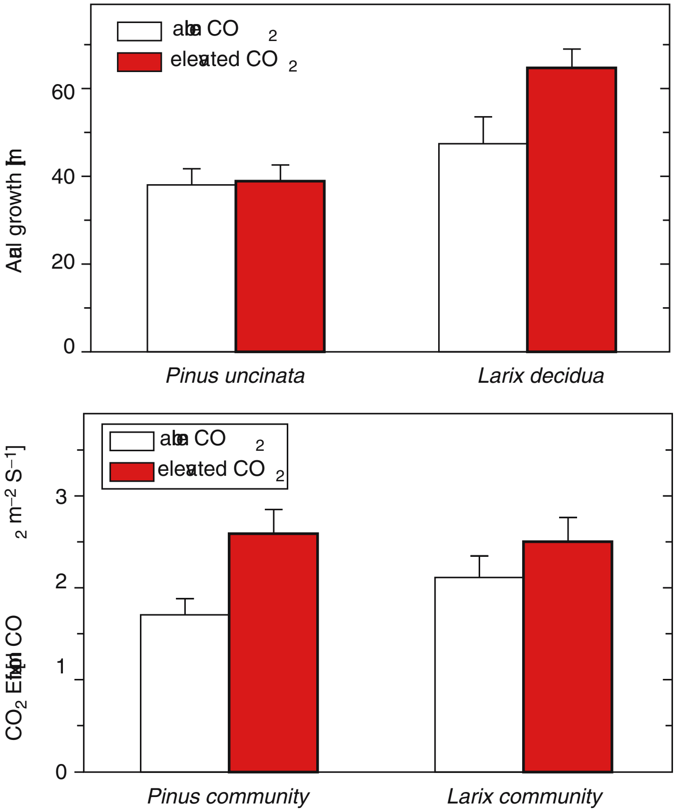


Fig. 6 Response of photosynthesis, growth and soil respiration to 5 years of CO_2 enrichment at the alpine treeline (plant data from Handa et al., 2005; soil data, F. Hagedorn, 2007). High altitude likely respond more sensitive to increasing atmospheric CO_2 than trees of the lowland since both gas exchange studies (Körner & Diemer, 1987) and global comparisons of stable carbon isotope data (Körner et al., 1991) show that the efficiency of CO_2 uptake at any given CO_2 partial pressure is higher in plants from high altitude. In a 5-year CO_2-enrichment study at the alpine treeline, Handa et al. (2005) found indeed that larch (Larix decidua) grew significantly better under elevated CO_2, while pine (Pinus uncinata) showed only a moderate initial response that disappeared thereafter. The growth response of both trees was significantly smaller than that of photosynthesis, indicating that the greater C-uptake was not used for a better growth

(several years) to higher temperatures (King et al., 2006). A long-term experiment in a boreal ecosystem has shown that soil warming overall stimulates decomposition processes more than site productivity and that an ecosystem in a warmer world can easily turn into a C source (Mack et al., 2004).

4.3 Snow Cover

Snow cover plays a major role in respiration losses of CO_2 from soils. Under an insulating snow cover, soil microbes are still active and very high concentrations in CO_2 can build up. These respiratory losses can diminish or even offset the C sequestration during the growing season. Important is that a continuous snow cover establishes before the soil temperature falls below a critical threshold temperature. The respiratory C losses from soils can be as high as 30–50% of the annual C fixation (Winston et al., 1997; Monson et al., 2006).

Values as high as 3,000 ppm CO_2 are commonly found, with a gradual decrease towards the snow surface (Körner, 2003; Mariko et al., 1994; Musselman et al., 2005).

The microbiology of CO_2 production from mountain forest soils during the winter is intriguing. The early concept was that soil microbes go into dormancy during winter and produce very little CO_2 (Tranquillini, 1979). Snow cover slows but not impedes soil microbial activity. Snow has, however, a dual effect. It limits growth because plant productivity commences when certain T threshold is exceeded, and it insulates the soil from cold temperatures (Peterson & Peterson, 2001). Only where the soil T drops below −7°C to −5°C, the rate of soil respiration becomes negligible (Brooks et al., 2004). Consequently, snow depth is a major determinant of microbial activity during winter and thus, of respiration losses (Monson et al., 2006). Recent microbiological research has shown that distinct soil microorganisms are active in winter and in summer. The microbial population under the snow cover holds a narrow ecological niche (Monson et al., 2006; Schadt et al., 2003). When these microorganisms run out of fresh substrate, they can even mineralize old C, as shown by ^{14}C data (Monson et al., 2006). Recent microbiological research has shown that distinct soil microorganisms are active in winter and in summer. The microbial population under the snow cover holds a narrow ecological niche (Monson et al., 2006; Schadt et al., 2003). When these microorganisms run out of fresh substrate, they can even mineralize old C, as shown by ^{14}C data (Monson et al., 2006).

A trend of CO_2 fluxes during the winter months was observed by Hubbard et al. (2005). Contrary to the simplifying belief of constant low microbial activity, a decline in respiration rates from the first snow cover to mid winter was observed. Late in the snowy season, but still on a snow covered landscape the respiration rate built up again. Relying only on few respiration measurements in winter is unlikely to yield useful estimates for soil respiration.

In a warmer world the snow cover develops later and melts earlier in a year. The effect bears the surprise that, somewhat counterintuitively, soils will become colder. Soils will be longer unprotected by an insulating snow cover. An increase in the number

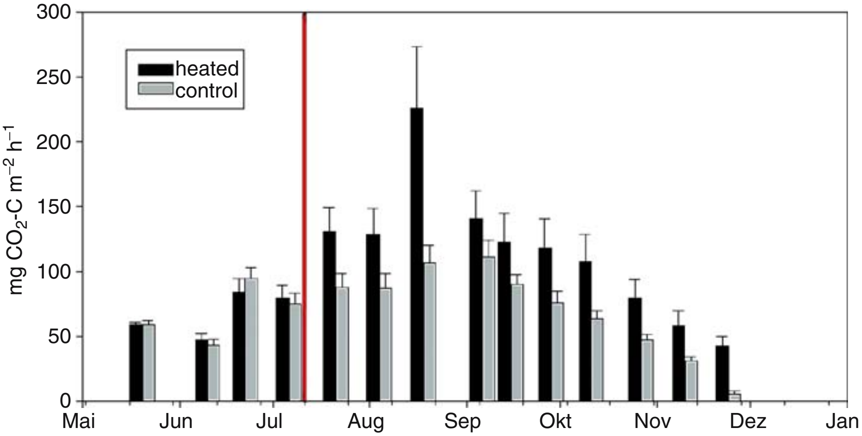


Fig. 7 Results of the soil warming experiment Achenkirch: Total soil respiration in a control plot and a heated plot at 3°C above ambient conditions

of freeze/thaw cycles resembles an ecosystem disturbance that can lead to N leaching and N_2O formation (Groffman et al., 2001).

Upon warming in spring respiration rates picks up with increasing temperatures. The ecosystem is split into an aboveground part that stays in dormancy under freezing conditions, and a reasonably warm soil that is insulated very well by a snow cover.

4.4 Fluxes of DOC

Dissolved organic carbon (DOC) is the mobile fraction of soil organic matter and it represents a key vehicle for the transport of nutrients in ecosystems (e.g. Hagedorn et al., 2000; Kalbitz et al., 2000). DOC is mobilized in the forest floor, the DOC flux into mineral soil is in the range of 5 to 50 g C m^{-2} $year^{-1}$ – a small but potentially important pathway of C loss (Michalzik et al., 2001; Neff & Asner, 2001). Sorption of DOC in the mineral soil significantly stabilizes SOM and hence the small DOC flux might contribute significantly to the accumulation and preservation of SOM in the long-term (Kaiser & Guggenberger, 2000). There are only a few studies looking at DOC-dynamics in mountain forest. Since the export of DOC from forest floor is controlled by C pools in the organic layer and water fluxes, DOC fluxes in mountain forests are likely to be in the upper range of ecosystems. In a subalpine spruce forest, Hagedorn et al. (2000) measured an export of 40 g DOC m^{-2} $year^{-1}$ from the forest floor and an export of 8.4 g DOC m^{-2} $year^{-1}$ from a subalpine catchment. Firstly, this illustrates that the major fraction of DOC is retained in the mineral soils, probably by sorption. Secondly, it shows that the DOC fluxes are at least in the same order as the fluxes from temperate ecosystems. In relative terms, DOC appears to be far more important for the overall C cycle in mountain than in low elevation forests, because the other soil C fluxes such as litterfall and heterotrophic respiration are smaller. In line with other C-fluxes in soils, DOC dynamics responds sensitive to any disturbance. After a moderate felling, Bäumler (1995) found a doubling of the DOC export from a subalpine catchment. In a boreal ecosystem the close link between the DOC flux and the amount of soil organic matter was identified. Increasing the productivity increases the annual litterfall and also the DOC flux. In terms of the C sequestration potential, an increase in growth rate does not necessarily lead to a proportional increase in the release of CO_2 from soils. Instead, soils can increase the pool of stable C, or c can be removed as DOC, and thus in a form that is not relevant for climatic change.

4.5 Nitrogen Oxides

The formation of N_2O in soils was long seen as a phenomenon of waterlogged soils (Bowden, 1986). Nitrogen limited ecosystems such as many mountain forests usually have a tight N-cycle. Currently, many mountain forests still carry the legacy of centuries of nutrient exploitation and are critically deprived in N (Führer, 2000). Consequently, N is withheld tightly and is still limiting biomass production. With the increase in N deposition, the demand of ecosystems may be at least temporarily exceeded and N can leave the system via the aquatic or the gaseous pathway (Perakis & Hedin, 2002; Vitousek et al., 1997). Evidence for nitrogen oxide (N_2O and NO_x) emissions from N enriched lowland forests has been given (Butterbach-Bahl et al., 2002). In recent experiments it was shown that also mountain forest soils can release relatively large quantities of nitrogen oxides (Härtel et al., 2002; Pilegaard, 2001) (Fig. 8). The annual release from a Rendzic Leptosol, however, was only 1 kg N_2O-N ha^{-1}. In a subalpine forest on calcareous Gleysols, N losses via denitrification amounted to 1.7 kg N/ha/a under ambient N deposition and 2.9 kg N ha^{-1} $year^{-1}$ under experimentally increased N inputs of 30 kg N ha^{-1} $year^{-1}$ (Kitzler et al., 2006; Mohn et al., 2000). These figures appear to be low, compared to the annual releases of 1.5–4 t CO_2-C ha^{-1}. Taking into account that N_2O is more effective than CO_2 as a GHG, than more than 10% of the annual GHG release comes from N_2O and N_2O consumes approx 20% of the annual C sequestration in the aboveground biomass. While the formation of N_2O is rather an issue on high-pH soils, on acidic soils N_2O emissions are accompanied by NO emissions (Venterea et al., 2003). The formation of N_2O is driven by the N availability and by temperature and is therefore a seasonal phenomenon. Given that N deposition remains high and that temperatures will increase indeed, the emission of nitrogen oxides from forest soils may become an issue of environmental concern (Schindlbacher & Zechmeister-Boltenstern, 2004). Mountain ranges tend to capture precipitation, especially when they form a prominent barrier in the landscape towards the prevailing wind direction. Even low concentrations of N in rain can add up to high amounts of N deposition which additionally promotes future emissions of nitrogen oxides from mountain forest soils.

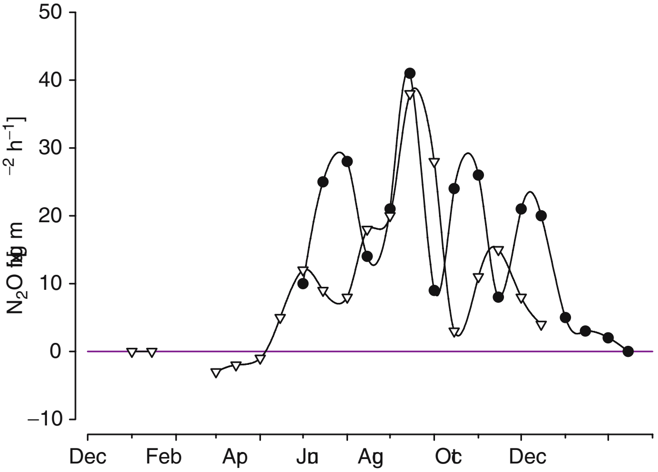


Fig. 8 Nitrous oxide (N_2O) formation in Achenkirch in 2 consecutive years (Härtel et al., 2002)

5 System Dynamics

In North America, Scandinavia and Siberia, treelines are shifting upwards indicating that the warming during the last century induces a large scale change in vegetation (Moiseev & Shiyatov, 2001; Kullmann, 2002). In the Alps, a declining use of alpine meadows lead to a reforestation and forest areas in are currently expanding. Upward shifting forest will clearly increase biomass C stocks, its effects on soil C are less clear, because this shift will not only increase C inputs into soils, but also increase soil temperatures and hence, respiration losses from soils. Sjögersten and Wookey (2002) found that tundra soils above the treeline contain more labile C than forest soils, suggesting that a warming and a rise in treeline would induce CO_2 losses from soils. Along altitudinal gradients in the Ural, however, soil C stocks increased with decreasing altitude (Fig. 9), suggesting that the increase in C inputs is greater than the losses via respiration.

Land-use change has an impact on forest structure. A survey of Swiss mountain forests has shown that in areas with pasture, the stand structure is rather heterogeneous. This effect is apparently depending on the presence of actively managing land owners (Mayer & Stöckli, 2005). Such forests are to a higher degree capable of protecting against avalanches and are therefore to a higher degree able to maintain a high C density in the biomass.

Forests in high elevation suffer damages from storms, pests, avalanches, and rockfall. Among these factors, the exposure to pests is affected by climatic change. Mountain forests are a difficult environment for bark beetles that are among the most aggressive deleterious organisms for Norway spruce. Their habitat is, among other factors, limited by thermal constraints (Schwerdtfeger, 1981). With increasing

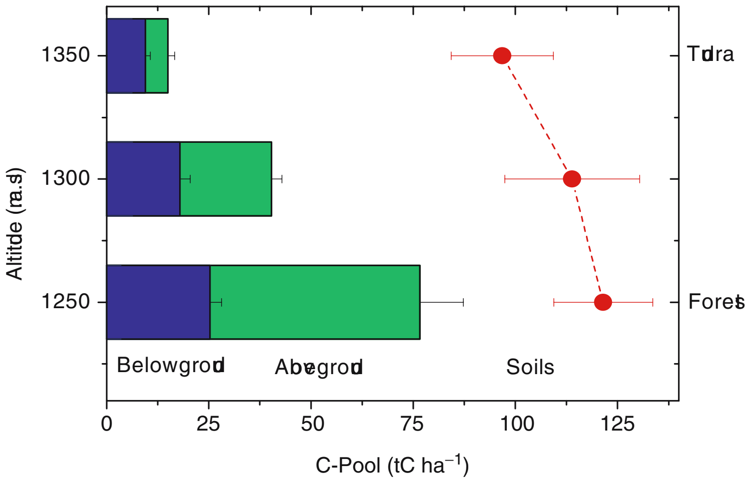


Fig. 9 Carbon (C) stocks along altitudinal gradients of the gentle slopes of the Southern Ural. Means and standard errors of five plots

temperatures these and other pest organisms are extending their habitat and the threat to mountain forests is increasing. The situation is aggravated, because forest managers are not necessarily expecting that forests in high elevation are endangered. Forest management costs are increasing, because beetle monitoring has to be extended into high elevation forests.

6 Conclusions

Climatic change will lead to an increase in the aboveground biomass. Due to the low overall productivity of mountain forests the rate of the incorporation of CO_2 in the biomass is slow. On the stand level, an overall increase in the C and N pool is expected. The effect is attenuated by an increase in the forest area as a consequence of a rising timberline and the encroachment or afforestation of alpine pastures. For an assessment of the regional effect, conclusions from the stand level need to be evaluated with respect of the disturbance regime. A reduction or constancy of the disturbance probability increases the potential of mitigating climatic change by elevated growth rates. If ecosystem disturbance (storm, pest infestation and others) increases, the mitigation power of forests decreases.

Whereas the productivity of mountain forests benefits from climatic change, the soils behave differently. Mountain forest soil store large amounts of C and N. The combination of elevated temperatures, the elongation of the growing season, and N-enrichment accelerate the mobilization of soil organic matter. The formation of GHGs in soils reduces the C sequestration effect. As long as climatic change allows the expansion of productive forests it is expected that mountain forests continue to be sinks of GHGs. Even modest increases in the soil temperature can increase the CO_2 emission by 20–30%. It remains to be shown in long-term ecological research programmes how long this ecosystem response prevails and at what level a new equilibrium status between climatic conditions and soil C and N pools will be reached.

References

Bäumler, R. (1995). Dynamik gelöster Stoffe in verschiedenen Kompartimenten kleiner Wassereinzugsgebiete in der Flyschzone der Bayrischen Alpen – Auswirkungen eines geregelten forstlichen Eingriffs. Bayreuther Bodenkundliche Berichte *40*, 133 p.

Beniston, M. (2006). Mountain weather and climate: A general overview and a focus on climatic change in the Alps. Hydrobiologia *562*, 3–16.

Bowden, W. B. (1986). Gaseous nitrogen emissions from undisturbed terrestrial ecosystems: An assessment of their impacts on local and global nitrogen budgets. Biogeochemistry ***2***, 249–279.

Broggi, M. F. (2002). Alpine space: A social challenge for research. Centralblatt für das gesamte Forstwesen *119* (3/4), 247–254.

Brooks, P. D., D. McKnight, K. Elder (2004). Carbon limitation of soil respiration under winter snowpacks: Potential feedbacks between growing season and winter carbon fluxes. Global Change Biology *11*, 231–238.

Butterbach-Bahl, K., R. Gasche, G. Willibald, H. Papen (2002). Exchange of N-gases at the Höglwald forest - a summary. Plant and Soil *240*, 117–123.

Carey, E. V., A. Sala, R. Keane, R. M. Callaway (2001). Are old forests underestimated as global carbon sinks? Global Change Biology *7*, 339–344.

Cernusca, A., P. Decker (1989). Faktorenabhängigkeit der respiratorischen Kohlenstoffverluste einer alpinen Grasheide (Caricetum curvulae) in 2300 m MH in den Hohen Tauern. In Cernusca, A. (ed.). *Struktur und Funktion von Graslandökosystemen im Nationalpark Hohe Tauern*. Universitätsverlag Wagner, Innsbruck, 371–396.

Davidson, E. A., I. A Janssens (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature *440*, 165–173.

De Gonzalo Aranoa, C. (2006). Efecto de la temperatura en la emisión de gases de efecto invernadero en suelos forestales del Tirol (Austria). Proyecto Fin de Carrera, Universidad Politécnica de Madrid.

Dullinger, S., T. Dirnböck, G. Grabherr (2004). Modelling climate change-driven treeline shifts: Relative effects of temperature increase, dispersal and invasibility. Journal of Ecology *92*, 241–252.

Falge, E., D. Baldocchi, J. Tennhunen (2002). Seasonality of ecosystem respiration and gross primary production as derived from FLUXNET measurements. Agricultural and Forest Meteorology *113*, 57–74.

Field, C. B., M. R. Raupach, R. Victoria (2004). The global carbon cycle: Integrating humans, climate, and the natural world. In Field, C. B., M. R. Raupach (eds.). *The global C cycle*. Scope Series 62, Island Press, Washington, DC.

Fröberg, M., D. Berggren Kleja, B. Bergkvist, E. Tipping, J. Mulder (2005). Dissolved organic carbon leaching from a coniferous forest floor - a field manipulation experiment. Biogeochemistry *75*, 271–287.

Führer, E. (2000). Forest functions, ecosystem stability and management. Forest Ecology and Management *132*, 29–38.

Grabherr, G., M. G. H. Pauli (1994). Climate effects on mountain plants. Nature *369*, 448.

Grace, J., M. Rayment (2000). Respiration in the balance. Nature *404*, 819–820.

Groffman, P. M., C. T. Driscoll, T. J. Fahey, J. P. Hardy, R. D. Fitzhugh G. L. Tierney (2001). Colder soils in a warmer world: A snow manipulation study in a northern hardwood forest ecosystem. Biogeochemistry *56*, 135–150.

Hagedorn, F., P. Schleppi, P. Waldner, H. Flühler (2000). Export of dissolved organic carbon and nitrogen from Gleysol dominated catchments - the significance of water flow paths. Biogeochemistry *50*, 137–161.

Handa, I. T., C. Körner, S. Hättenschwiler (2005). A test of the treeline carbon limitation hypothesis by in situ CO2 enrichment and defoliation. Ecology *86*,1288–1300.

Härtel, E., S. Zechmeister-Boltenstern, M. Gerzabek (2002). Gaseous nitrogen losses from a forest site in the North Tyrolean Limestone Alps. *Environmental Science and Pollution Research* (Special Issue 2), 23–30.

Havranek WM (1987) Physiologische Reaktionen auf Klimastreβ bei Bäumen an der Waldgrenze. GSF Bericht *10*, 23–36.

Hubbard, R. M., M. G. Ryan, K. Elder, C. C. Rhoades (2005). Seasonal patterns in soil surface CO2 flux under snow cover in 50 and 300 year old subalpine forests. Biogeochemistry *73*, 93–107.

Hungate, B. A., J. S. Dukes, M. R. Shaw, Y. Luo, C. B. Field (2003). Nitrogen and Climate Change. Science *302*, 1512–1513.

Huxman, T. E., A. A. Turnipseed, J. P. Sparks, P. C. Harley, R. K. Monson (2003). Temperature as a control over ecosystem CO2 fluxes in a high-elevation, subalpine forest. Oecologia *134*, 537–546.

IPCC (Intergovernmental Panel on Climate Change). (2001). *Climate change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, New York.

IPCC (Intergovernmental Panel on Climate Change). (2007). Summary for policymakers. In Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, H. L. Miller (eds.). *Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge.

Jandl, R., M. Lindner, L. Vesterdal, B. Bauwens, R. Baritz, F. Hagedorn, D. W. Johnson, K. Minkkinen, K. A. Byrne (2006). How strongly can forest management influence soil carbon? Geoderma *137* (3–4), 253–268.

Janssens, I. A., A. Freibauer, P. Ciais, P. Smith, G.-J. Nabuurs, G. Folberth, B. Schlamadinger, R. W. A. Hutjes, R. Ceulemans, E.-D. Schulze, R. Valentini, A. J. Dolman (2003). Europe's terrestrial biosphere absorbs 7 to 12% of European anthropogenic CO2 emissions. Science *300*, 1538–1542.

Jarvis, P., S. Linder (2000). Constraints to growth of boreal forests. Nature *405*, 904–905.

Kaiser, K., G. Guggenberger (2000). The role of DOM sorption to mineral surfaces in the preservation of organic matter in soils. Organic Geochemistry *31*, 711–725.

Kalbitz, K., S. Solinger, J.-H. Park, B. Michalzik, E. Matzner (2000). Controls on the dynamics of dissolved organic carbon in soils: A review. Soil Science *165*, 277–304.

King, A. W., C. A. Gunderson, W. M. Post, D. J. Weston, S. D. Wullschleger (2006). Plant respiration in a warmer world. Science *312*, 536–537.

Kitzler, B., S. Zechmeister-Boltenstern, C. Holtermann, U. Skiba, K. Butterbach-Bahl (2006). Controls over N2O, NOx and CO2 fluxes in a calcareous mountain forest. Biogeosciences *3*, 293–310.

Körner, C., M. Diemer (1987). In situ photosynthetic responses to light, temperature and carbon dioxide in herbaceous plants from low and high altitude. Functional Ecology *1*, 179–194.

Körner, C., G. D. Farquhar, S. C. Wong (1991). Carbon isotope discrimination by plants follows latitudinal and altitudinal trends. Oecologia *88*, 30–40.

Körner, C. (1998). A re-assessment of high elevation treeline positions and their explanation. Oecologia *115*, 445–459.

Körner, C. (2003). *Alpine plant life - functional ecology of high mountain ecosystems*. Springer, Berlin.

Kullmann, L. (2002). Rapid recent range-margin rise of tree and shrub species in the Swedish Scandes. Journal of Ecology *90*, 68–77.

Loya, W., P. Grogan (2004). Global change: Carbon conundrum on the tundra. Nature *431*, 406–408.

Mack, M. C., E. A. G. Schuur, M. S. Bret-Harte, G. R. Shaver, F. S. Chapin (2004). Ecosystem carbon storage in arctic tundra reduced by long-term nutrient fertilization. Nature *431*, 440–443.

Matulla, C., E. K Penlap, P. Hass, H. Formayer (2003). Comparative analysis of spatial and seasonal variability: Austrian precipitation during the 20th century. International Journal of Climatology *23*, 1577–1588.

Mariko, S., Y. Bekku, H. Koizumi (1994). Efflux of carbon dioxide from snow-covered forest floors. Ecological Research *9*, 343–350.

Mayer, A. C., V. Stöckli (2005). Long-term impact of cattle grazing on subalpine forest development and efficiency of snow avalanche protection. Arctic, Antarctic, and Alpine Research *37* (4), 521–526.

Melillo, J. M., P. A. Steudler, J. D. Aber, K. Newkirk, H. Lux, F. P. Bowles, C. Catricala, A. Magill, T. Ahrens, S. Morrisseau (2002). Soil warming and carbon-cycle. Feedbacks to the climate system. Science *298*, 2173–2176.

Michalzik, B., K. Kalbitz, J.-H. Park, S. Solinger, E. Matzner (2001). Fluxes and concentrations of dissolved organic carbon and nitrogen - a synthesis for temperate forests. Biogeochemistry *52* (2), 173–205.

Mohn, J., A. Schurmann, F. Hagedorn, P. Schleppi, R. Bachofen (2000). Increased rates of denitrification in nitrogen-treated forest soils. Forest Ecology and Management *137*, 113–119.

Moiseev, P. A., S. G. Shiyatov (2001). Vegetation dynamics at the treeline ecotone in the Ural highlands, Russia. In Nagy, L., G. Grabherr, C. Körner (eds.). *Alpine biodiversity in Europe*. Ecological Studies 167. Springer, Berlin, pp. 423–435.

Monson, R. K., A. A. Turnipseed, J. P Sparks, P. C. Harley, L. E. Scott-Detton, K. Sparks, T. E. Huxman (2002). Carbon sequestration in a high-elevation, subalpine forest. Global Change Biology *8*, 459–478.

Monson, R. K., D. L. Lipson, S. P. Burns, A. A. Turnipseed, A. C. Delany, M. W. Williams, S. K. Schmidt (2006). Winter forest soil respiration controlled by climate and microbial community composition. Nature *439*, 711–714.

Musselman, R. C., W. J. Massman, J. M. Frank, J. L. Korfmacher (2005). The temporal dynamics of carbon dioxide under snow in a high elevation Rocky Mountain subalpine forest and meadow. Arctic, Antarctic, and Alpine Research *37* (4), 527–538.

Neff, J. C., G. P. Asner (2001). Dissolved organic carbon in terrestrial ecosystems: Synthesis and a model. Ecosystems *4*, 29–48.

Nicolussi, K., Kaufmann, M., Patzelt, G., van der Plicht, J., Thurne, A. (2005). Holocene tree-line variability in the Kauner Valley, Central Eastern Alps, indicated by dendrochronological analysis of living trees and subfossil logs. Vegetation History and Archaeobotany *14* (3), 221–234.

Oren, R., D. S. Ellsworth, K. H. Johnsen, N. Phillips, B. E. Ewers, C. Maier, K. V. R. Schäfer, H. McCarthy, G. Hendrey, S. G. McNulty, G. G. Katul (2001). Soil fertility limits carbon sequestration by forest ecosystems in a CO2-enriched atmosphere. Nature *411*, 469–472.

Rayn, M. G. (1991). Effects of climate change on plant respiration. Ecological Applications *1*, 157–167.

Paulsen, J., U. M. Weber, C. Körner (2000). Tree growth near treeline: Abrupt or gradual reduction with altitude? Arctic Antarctic and Alpine Research *32*, 14–20.

Perakis, S. S., L. O. Hedin (2002). Nitrogen loss from unpolluted South American forests mainly via dissolved organic compounds. Nature *415*, 416–419.

Perruchoud, D., F. Kienast, E. Kaufmann, U. O. Bäcker (1999). 20th century carbon budget of forest soils in the Alps. Ecosystems *2*, 320–337.

Peterson, D. W., D. L. Peterson (2001). Mountain hemlock growth responds to climatic variability at annual and decadal time scales. Ecology *82* (12), 3330–3345.

Pilegaard, K. (2001) Air-soil exchange of NO, NO2 and O3 in forests. Water, Air, and Soil Pollution Focus *1*, 79–88.

Rodeghiero, M., A. Cescatti (2005). Main determinants of forest soil respiration along an elevation/temperature gradient in the Italian Alps. Global Change Biology *11*, 1024–1041.

Schadt, C. W., A. P. Martin, D. A Lipson, S. K. Schmidt (2003). Seasonal dynamics of previously unknown fungal lineages in tundra soils. Science *301*, 1359–1361.

Schindlbacher, A., S. Zechmeister-Boltenstern (2004). Effects of soil moisture and temperature on NO_2, NO2, and N_2O emissions from European forest soils. Journal of Geophysical Research *109* (D17302), 1–12.

Schlesinger, W. H., J. A. Andrews (2000). Soil respiration and the global carbon cycle. Biogeochemistry *48*, 7–20.

Schwerdtfeger, F. (1981). *Die Waldkrankheiten. Ein Lehrbuch der Forstpathologie und des Forstschutzes*. Verlag Paul Parey, Hamburg.

Shaver, G., J. Canadell, F. Chapin, J. Gurevitch, J. Harte, G. Henry, P. Ineson, S. Jonasson, J. Melillo, L. Pitelka, L. Rustad (2000). Global warming and terrestrial ecosystems: A conceptual framework for analysis. BioScience *50*, 871–882.

Siegwolf, R. (1987). CO2-Gaswechsel von Rhododendron ferrugineum L. im Jahresgang an der alpinen Waldgrenze. Ph.D. thesis, Institute of Botany, University of Innsbruck.

Sjögersten, S., P. A. Wookey (2002). Climatic and resource quality controls on soil respiration across a forest- tundra ecotone in Swedish Lapland. Soil Biology and Biochemistry *34*, 1633–1646.

Spiecker, H., K. Mielikäinen, M. Köhl, J. P. Skovsgaard (1996). *Growth trends in Europe - studies from 12 countries*. EFI Research Reports 5. Springer, Heidelberg.

Strömgren, M., S. Linder (2002). Effects of nutrition and soil warming on stemwood production in a boreal Norway spruce stand. Global Change Biology *8*, 1195–1204.

Tranquillini, W. (1979). *Ecological physiological ecology of the alpine timberline - tree existence at high altitude with special reference to the European Alps*. Studies 31. Springer, New York.

Valentini, R., G. Matteucci, A. J. Dolman, E.-D. Schulze, C. Rebmann, E. J. Moors, A. Granier, P. Gross, N. O. Jensen, K. Pilegaard, A. Lindroth, A. Grelle, C. Bernhofer, T. Grünwald, M. Aubinet, R. Ceulemans, A. S. Kowalski, T. Vesala, Ü. Rannik, P. Berbigier, D. Loustau, J. Guõmundsson, H. Thorgeirsson, A. Ibrom, K. Morgenstern, R. Clement, J. Moncrieff, L. Montagnani, S. Minerbi, P. G. Jarvis (2000). Respiration as the main determinant of carbon balance in European forests. Nature *404*, 861–865.

Venterea, R. T., P. M. Groffman, L.V. Verchot, A. H. Magill, J. D. Aber, P. A. Streudler (2003). Nitrogen oxide gas emissions from temperate forest soils receiving long-term nitrogen inputs. Global Change Biology *9*, 346–357.

Vetter, M., C. Wirth, H. Böttcher, G. Churkina, E.-D. Schulze, T. Wutzler, G. Weber (2005). Partitioning direct and indirect human-induced effects on carbon sequestration of managed coniferous forests using model simulations and forest inventories. Global Change Biology *11*, 810–827.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger, D. G. Tilman (1997). Human alteration of the global nitrogen cycle: Sources and consequences. Ecological Applications *7*, 737–750.

Wieser, G. (2004a). Environmental control of carbon dioxide gas exchange in needles of a mature Pinus cembra tree at the alpine timberline during the growing season. Phyton *44*, 145–153.

Wieser, G. (2004b). Seasonal variation of soil respiration in a Pinus cembra forest at the upper timberline in the Central Austrian Alps. Tree Physiology *24*, 475–480.

Wieser, G., M. Bahn (2004). Seasonal and spatial variation of woody tissue respiration in a Pinus cembra tree at the alpine timberline in the central Austrian Alps. Trees *18*, 576–580.

Wieser, G., D. Stöhr (2005). Net ecosystem carbon dioxide exchange dynamics in a Pinus cembra forest at the upper timberline in the central Austrian Alps. Phyton *45*, 233–242.

Wieser, G., M. Tausz (2006). Trees at their upper limit. Treelife limitation at the Alpine Timberline. *Plant ecophysiology*, Vol 5. Springer, Berlin.

Winston, G. C., E. T. Sundquist, B. B. Stephens, S. E. Trumbore (1997). Winter CO2 fluxes in a boreal forest. Journal of Geophysical Research *102* (D24), 28795–28804.

Zeller, K. F., Nikolov, N. T. (2000). Quantifying simultaneous fluxes of ozone, carbon dioxide and water vapor in a subalpine forest. Environmental Pollution *107*, 1–20.

Zha, T., S. Kellomäki, K. Y. Wang, I. Rouvinen (2004). Carbon sequestration and net ecosystem respiration for 4 years in a Scots pine forest. Global Change Biology *10*, 1492–1503.

Part III
Monitoring and Modeling

Estimating Carbon Stocks and Stock Changes in Forests: Linking Models and Data Across Scales

V. LeMay[1]* and W. A. Kurz[2]

1 Introduction

Concerns over the impacts of atmospheric changes on the global climate system have resulted in a global emphasis on altering anthropogenic activities to reduce the rate of atmospheric change. The atmospheric concentration of carbon dioxide (CO_2) rose from 280 ppm prior to the industrial revolution to ~380 ppm in 2005 and other greenhouse gases have also increased (e.g., methane from 715 to 1,774 ppb[1]) (IPCC, 2007). Increases in the amount of atmospheric carbon have been linked to changes in climate, including a 0.74°C temperature increase in the last 100 years (IPCC, 2007). Christensen et al. (2007) note that all land regions will likely warm in the 21st century.

Efforts to reduce the amount of carbon and other emissions are being made, as there is increasing evidence that this warming, particularly over the past 50 years, is partly attributable to human activities. The United Nations Framework Convention on Climate Change (UNFCCC) has been developed to initiate global action including examining the causes and magnitudes of carbon sinks and sources, with a view to possibly increase carbon uptake or reduce carbon losses through management. As part of this initiative, monitoring and annual reporting of emissions and removals is one of the commitments made by parties to the UNFCCC.

Forests have been identified as possible sinks that may offset emissions produced by burning fossil fuels (e.g., Myneni et al., 2001; Binkley et al., 2002; Vågen et al., 2005). In forests, carbon is accrued through increased live biomass and/or increased dead organic matter and soil carbon, whereas carbon is released to the atmosphere through respiration, decomposition, and burning. Harvest transfers result in subsequent release of carbon from decomposition during wood processing and decay of

---

[1] Professor with the Forest Resources Management Department, University of British Columbia, 2045-2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada, * Valerie. LeMay@ubc.ca

[2] Senior Scientist with Natural Resources Canada, Canadian Forest Service, 506 West Burnside Road, Victoria, BC, V8Z 1M5, Canada, wkurz@nrcan.gc.ca

[1] http://www.unfccc.de; accessed February 14, 2006.

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

wood products in use or in landfills. Harvest also transfers biomass to dead organic matter (slash) in the forest from where carbon will be released through decomposition. Activities such as afforestation and reforestation promote the extent of forests, whereas fire, harvest, disease, and insects reduce live forest biomass. Nabuurs et al. (2007) concluded that the expected carbon mitigation benefits of reducing deforestation would be greater than the benefits of afforestation, in the short term.

Procedures to estimate the amount of carbon in forests are sometimes labeled as "carbon budgeting". Estimates of past and current forest carbon stocks are needed, along with future projections and estimates of possible sequestration increases through forest management activities such as afforestation, reforestation, stand management, and forest protection. Countries that have committed to the Kyoto Protocol will need to provide detailed reports on land use, land-use change, and forestry (LULUCF) activities (Penman et al., 2003). Although reporting may be for a large land area (e.g., country-wide), more localized information will be needed to detect changes and related causes (land transition matrix), and to forecast effects of management activities. These reporting needs have prompted a variety of methodological approaches to integrate a number of data sources and models at a variety of spatial and temporal scales into a reporting system.

In this paper, we present a discussion of methods used to obtain information on carbon stocks, using reported analyses as examples. We begin with some of the challenges in obtaining the necessary information. Methods used to integrate data and models across time and spatial scales, including stratification into large ecosystems, and imputation and regression methods to expand data to unsampled locations, are then discussed. In the summary, we list some of the research needs for improving these linkages and resulting estimates. As there are many articles relating to carbon budgeting, a full review is not given. Instead, examples of research, weighted to more current articles, are referenced.

2 Monitoring Challenges

Monitoring forests for changes in carbon utilizes many of the methods developed for forest inventory of any variable of interest (e.g., forest growth and yield). However, monitoring for carbon introduces additional complexity over commonly measured forest inventory variables, since below-ground components (e.g., ephemeral fine roots, coarse roots, soil carbon) and additional above-ground components, including litter and dead organic matter (IPCC, 2007), are of interest (Fig. 1).

Many of these components are spatially very variable, resulting in difficulties in measuring carbon (e.g., soil carbon). Ecosystem carbon pools such as dead organic matter, litter and soil carbon are typically not included in conventional forest inventories. The process of greenhouse gas exchange is more difficult to estimate and model than tree growth. Further, emissions of non-CO_2 greenhouse gasses associated with wildfires, forested wetlands, and forest management (e.g. nitrogen fertilization) are difficult to quantify, and are also a reporting requirement.

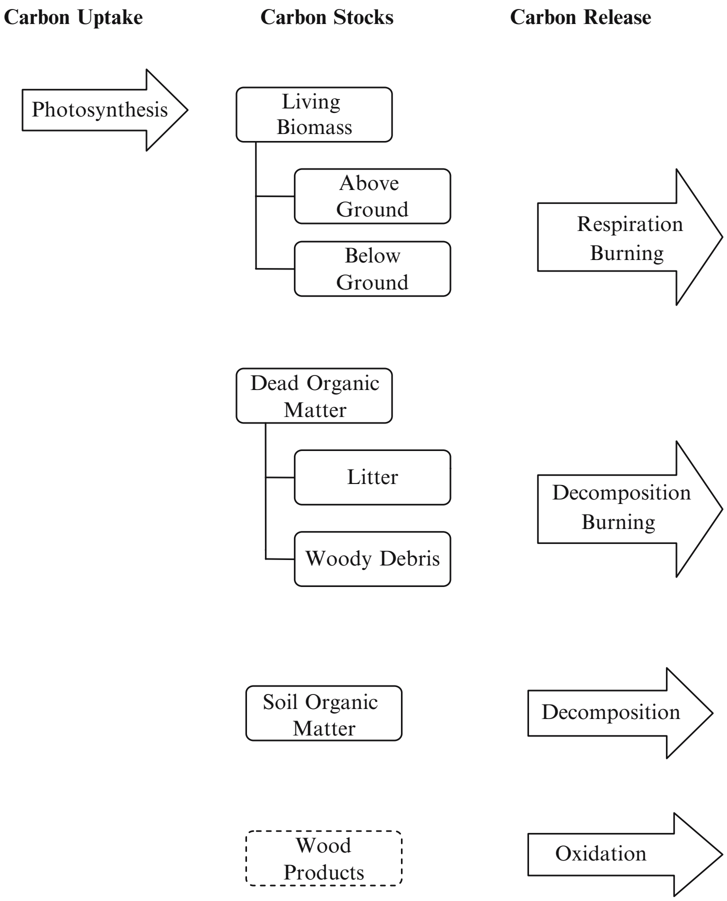


Fig. 1 Breakdown of carbon stocks and processes that result in changes to carbon stocks

Although there are unique circumstances in each administrative unit, regardless of size of that unit, the process of obtaining estimates of past, current, and future forest carbon stocks generally involves:

1. Identifying the current extent of managed forest land
2. Estimating the changes, additions (e.g., afforestation, encroachment) and deletions (e.g., deforestation such as conversion to urban areas) to the forest land base that have occurred during the monitoring period (forest land transition matrix)
3. Estimating the carbon stocks and stock changes for all managed forest land, including:
   (a) Above- and belowground biomass, including stems, foliage, tops and branches and fine and coarse roots
   (b) Dead organic matter (litter and dead wood) and soil organic carbon

4. Forecasting future carbon/biomass, given
   (a) Natural ecosystem dynamics and disturbances
   (b) Human-caused disturbances, including: harvesting, reforestation, afforestation, change to other uses (e.g., urban, agriculture) and insect/disease management

As well as estimating past and current carbon stocks, the reasons for these changes are needed. This information can be used to assess the impacts of past management activities, and to recommend future forest management activities that may increase sequestration and decrease carbon release.

2.1 Extent of Managed Forest Land

The definition of forest land given by FAO, by UNFCCC, and under the Kyoto Protocol differ. For example, using some definitions for numbers or areas covered by trees, cities with extensive urban tree cover, such as Vancouver, Canada, would be largely classified as forest land. Also, defining the forest land boundary is not always clear, the definition of stand edge has also been disputed (Luken et al., 1991). Because some definitions of forest include recently disturbed forests that are expected to revert back to forest, knowledge of current land cover alone may be insufficient for land classification. Forest lands designated for conservation, harvest, water protection, etc., can all be part of the managed forest and need to be included in the monitoring process.

2.2 Forest Land Transition Matrix

Information on changes in forest land area and distribution by type (forest land transition matrix), and the causes for these changes are needed. The land transition matrix describes the area remaining in each land category and the area transitions between land categories resulting from human activities such as afforestation, reforestation, deforestation and other land-use changes. In addition, information is required on the changes in carbon stocks due to natural disturbances and forest management. Obtaining information on forest changes due to human and/or natural disturbance involves gathering information on a variety of scales, often from a number of agencies, including private companies and land owners.

2.2.1 Human Disturbances

Areas that are more populated are likely to have a large diversity of human disturbances including conversions from forest to agriculture, mining, infrastructure development, or urban use (and vice versa). Under the UNFCCC and the Kyoto

Protocol, changes in land type through land management activities must be reported separately from other human activities. Obtaining information on these human-caused land-use changes involving forests involves gathering and merging information on a variety of scales, often from a number of agencies, including private companies and land owners (e.g., Mouillet & Field, 2005; White & Kurz, 2005). This information is needed to separate deforestation losses, due to urbanization and other land use changes, from temporary loss in forest cover due to natural causes such as wildfire or forest harvest.

2.2.2 Forest Management

Forest management activities from site preparation to stand tending to final harvest all affect forest carbon dynamics. Information on the area affected by forest management activities and on the magnitude of the management impacts on carbon stocks is required. Recent policy shifts towards continuous forest cover management, have led to the more common practice of partial removals for extracting timber, including removals of single trees, and regularly and irregularly shaped groups of trees that are difficult to monitor. Delineating boundaries on any land change is often very difficult, even when the change is dramatic, such as with clearcutting or conversion to urban land, but this is especially difficult for small-scale disturbances and partially harvested stands.

2.2.3 Natural Disturbances

In many forests of the world, natural disturbances play a dominant role in forest carbon dynamics (e.g., Kurz & Apps, 1999; Li et al., 2003b; Mouillet & Field, 2005). Catastrophic natural disturbances, such as fires or epidemic insect outbreaks, are more visible, whereas regular smaller-scale disturbances, such as windthrow and landslides are harder to monitor, but may be of regional significance. Encroachment of trees into non-treed areas can be a very slow process, but may occur along forest edges (e.g., alpine tree-line, forest/grassland edges, etc.). When these areas of forest change are in remote, sparsely populated areas, monitoring can be more of a challenge, as these events are not detected unless a formal, regular monitoring program has been implemented.

2.2.4 Stand Dynamics

In addition to changes in forest land due to disturbances, change due to stand dynamics must be monitored. Stands may have been disturbed and regenerated during the monitoring period. Also, changes in species composition, density, and tree sizes, particularly for fast-growing forests, will impact the forest carbon dynamics. For example, areas that are primarily deciduous (hardwood) at the beginning of the monitoring period, may be mixed deciduous/coniferous at the end of the period through successional changes. These changes in stands will be associated with changes in other components, such as litter, woody debris (e.g., Ganjegunte et al., 2004), etc., associated with these changes in above-ground live biomass.

2.3 Estimates of Carbon Stocks and Stock Changes

Estimates of carbon stock and stock changes at the national, regional or landscape-scale for the managed forest are needed. Individual forest stands can be carbon sources or sinks, depending on the stage of stand development (e.g., Gower et al., 1996). Shortly after disturbance, the rate of carbon uptake in growing trees can be less than the carbon loss from decaying slash and other dead organic matter. Vigorously growing stands, such as young even-aged stands, will have higher rates of carbon uptake, while stands with older trees, take up carbon at lower rates, but store the largest quantities of carbon per hectare.

Where detailed inventory information is available at the beginning and the end of the monitoring period, the average annual carbon stock changes can be estimated by calculating the difference in carbon stocks divided by the number of years in the monitoring period. Care must be taken, however, to not confound the estimates of carbon stock changes with changes in managed forest area. Alternatively, a single estimate of carbon stocks, for example at the beginning of the monitoring period, combined with detailed information on forest change, such as tree growth, dead organic matter and soil carbon dynamics, and changes resulting from land-use change, forest management and natural disturbances can be used to estimate annual changes (termed a “carbon budget” model). The advantage of this second approach is that detailed annual estimates of carbon stock changes can be provided, including an account of the inter-annual variability brought upon by variations in harvest rates or natural disturbances (Kurz & Apps, 1999).

For smaller land areas and for experiments, carbon components may be measured directly (e.g., Yang et al., 2005) for each time period. However, for larger land areas, direct field measures are often not possible (Li et al., 2002). Instead models are used to estimate some or all of the carbon components from variables, measured or imputed, over the entire land area (e.g., Beets et al., 1999; Li et al., 2002, 2003a). Some components, particularly above-ground carbon from live biomass, are easier to measure and/or estimate from commonly used forest inventory measures. Biomass estimates are commonly obtained from other measured variables (e.g., tree size, stand merchantable volume, vegetation indices from remotely sensed data) obtained from a sample, in the case of ground-measured variables, or the complete coverage of the forested area, in the case of remotely-sensed variables. Fang et al. (2001) used a carbon expansion factor to estimate above-ground tree carbon from merchantable timber volume. Myneni et al. (2001) used the normalized difference vegetation index (NDVI) and National Oceanic and Atmospheric Administration series satellites 7, 9, 11, and 14 coupled with ground data (stem wood volume) to estimate the biomass and used this to estimate change in terrestrial carbon storage and sinks in Northern (hemisphere) forests. The biomass equations are often critical to carbon estimation (Rohner & Böswald, 2001), since they often are the linkage between forest inventory measures, and carbon stock estimates. Due to the cost and destructive nature of data collection to develop biomass equations, these equations are often developed for large land areas. Jenkins et al. (2003) used a meta-modelling approach with previously developed equations for many locations and species, to develop country-wide hardwood versus softwood biomass equations for the US, for example. They noted that these would not be accurate for lower spatial scales, however. For areas with many species, such as tropical forests, development of volume, and therefore, biomass, from size or other inventory measures is difficult (Akindele & LeMay, 2006). Chave et al. (2005) used an extensive database to develop biomass equations for tropical species by grouping all species into broad forest types.

Other more difficult to measure components, including below-ground components, may be estimated from the above-ground live biomass in a system of estimating equations, rather than measured directly over the forest land area (e.g., Kurz et al., 1999; Cairns et al., 1997; Li et al., 2003a). Bi et al. (2004) discuss the issues in obtaining additive biomass components and recommend estimating the set of equations as a system to obtain logical consistency, and to improve estimating efficiency, the approach later used by Lambert et al. (2005). Estimating dead organic matter, litter, and soil carbon pools is much more difficult, since the amount of carbon in these pools is affected by the current vegetation, the time and type of last disturbance, and other site and climatic factors (e.g., Ganjegunte et al., 2004). Approaches based on correlation with current vegetation have had limited success. Smith and Heath (2002) used a meta-modelling approach to summarize information on forest floor carbon mass for the US. Alternative approaches involving detailed vegetation and site analyses or modelling of past disturbance history and stand dynamics (e.g., Kurz & Apps, 1999) are being developed and refined.

2.4 Forecasting Future Carbon/Biomass

Estimates of future carbon stocks are of interest to evaluate management alternatives or policy options. Nelson (2003) noted that models to obtain landscape level information are often difficult to explain, since the number of model components is often great and analysis is difficult to conduct and repeat. Also, the use of different models can lead to different results. For example, Nuutinen and Kellomäki (2001) present results using three different models to estimate timber production and carbon components. For short time periods, good agreement between estimates of change may be obtained. However, for long time periods, there is greater uncertainty, both in terms of carbon dynamics, and disturbance types, frequency, and impacts.

Models of carbon uptake and release are frequently used in estimating net carbon storage (e.g., Kurz & Apps, 1999; Schimel et al., 2000), and may be used to forecast changes using assumptions about future rates of management activities and natural disturbances. Forested landscapes are carbon sources if the sum of the stand-level carbon stocks is decreasing, for example as the result of increases in harvest rates or rates of stand-replacing natural disturbances. Conversely, reduction in rates of natural disturbances, or lengthening of harvest rotations are changes that typically bring about landscape-level carbon sinks (Kurz et al., 1998). Approaches involving stand to landscape-level scaling of carbon stock changes are well suited for landscapes with uniform, even-aged stands. Estimating changes due to management practices will require detail for individual stands (i.e., similar species, age, height, etc. composition), or, groups of stands (strata/ecological units), and basic research to calibrate and/or create models (e.g., Ganjegunte et al., 2004; Li et al., 2003b; Oliver et al., 2004; Kurz et al.,2002; Yang et al., 2005). The complexity of harvest patterns has increased, even in temperate forests, and more complex stand structures and cutting patterns make estimation of carbon stock changes increasingly difficult. In tropical areas, the large species diversity increases the complexity of modeling carbon stock changes. Finally, predicting changes due to possible management regimes is complicated by the interactions between management impacts and expected changes in species ranges and growth rates due to climatic shifts (Cao & Woodward, 1998). Forecasting requires models that can reflect the stand (or strata) changes, and methods to scale these up to the larger land unit.

3 Integrating Multiple Data Sources Across Spatial Scales

There are many variations in approaches used to obtain information across spatial scales at one time period, or at more than one time to obtain change data. Following Zeide (2003), these can be loosely divided into:

1. “Bottom-up” approaches, based on an extensive network of repeatedly measured, telescoping ground plots (nested plots of decreasing sizes), that are aggregated to information for higher scales. In forest inventory literature, this is often termed a continuous forest inventory (CFI).
2. “Top-down” approaches, where remotely sensed data are connected to available inventory data and other remotely sensed images, via imputation, and spatial/non-spatial modeling. Information at lower scales is obtained via disaggregation.

Kauppi (2003) divided the approaches similarly using the terms “inventory” versus “non-inventory” approaches. A discussion of these two types of approaches is given here. However, in practice, a mixture of “top-down” and “bottom-up” approaches is often used, which Zeide (2003) termed the “U approach”.

3.1 “Bottom-Up” Approach

For any forest inventory statistic, the “bottom-up” approach involves a base of repeatedly measured plots (CFI) that are aggregated for higher spatial scales (Fig. 2).

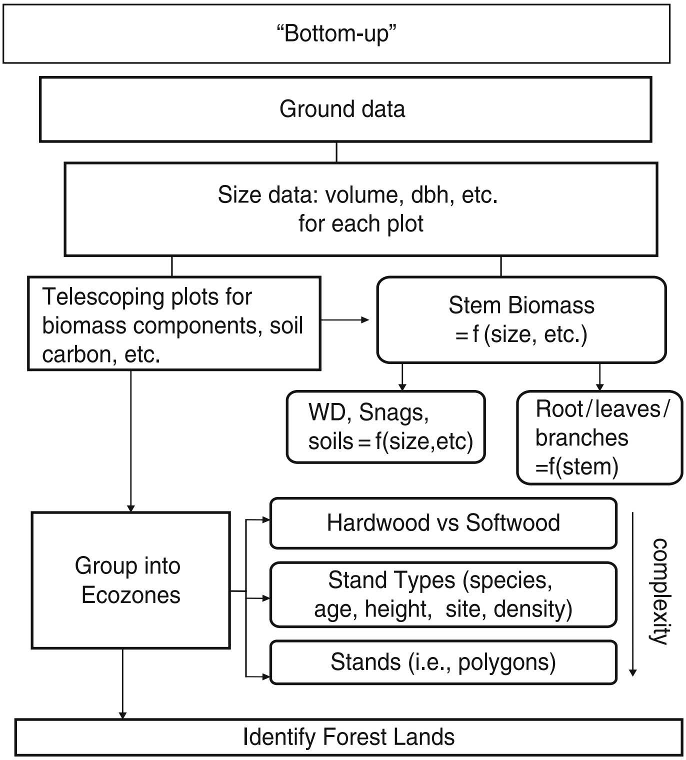


Fig. 2 "Bottom-up" approach beginning with an extensive network of telescoping plots, aggregated into strata, and used to delineate forest land where WD is woody debris

This approach is often very labor intensive, but may require lower technological costs.

The CFI approach is largely data driven, with commonly accepted methods for data analysis based on the inventory approach used. A number of desirable statistical properties are associated with a CFI approach, including:

1. The probability associated with each plot (subplot) is known for each inventory cycle. Unbiased estimates of each measured variable of interest (e.g., carbon stocks by stratum) can be obtained.
2. Relationships across the scales of measurement represented by the telescoping plots can be examined and used in estimation. For example, soil carbon can be related to overstory stand characteristics.
3. Variance estimates are also easy to obtain for variables that are directly measured. For variables that are estimated from measured variables (e.g., biomass estimated from tree size measures), variance estimates can be calculated, but commonly model estimates are treated as true measures with no variance.
4. Since the CFI is repeated over time, estimates of the forest land transition matrix, and reasons for these changes are unbiased and more precise than using separate inventories at each time (Cochran, 1977).

The results are then less disputed, and repeatable by different users of the data (e.g., Kauppi, 2003). Methods used are often easier to explain in fairly simple terms, increasing the trust and confidence in the estimates. As part of the inventory process, information on current land use can be gathered.

Commonly, a systematic layout of ground plots is used in a CFI. Post-stratification is easily implemented, and can be changed over time. For example, China used a national forest inventory system of ground plots to estimate forest area and other statistics at provincial levels (Fang et al., 2001). At each plot location, telescoping plots (i.e., nested plots) can be used to efficiently measure live biomass, dead organic matter, and soil carbon pools. Larger plots can be used for larger-scale components (e.g., stem biomass) whereas smaller plots can be used for smaller-scale components (e.g., soil carbon and fine roots). Consistency can be maintained across spatial scales, since all estimates could be based on the same network of plot data.

A major disadvantage to the "bottom-up" approach can be the cost of obtaining the information to achieve a desired level of precision. Few plots are needed to obtain desired precision for the larger-scale components, whereas many plots may be needed for smaller-scale components since the between-plot variability is much greater. Where labor costs are very low or the managed forest land area is smaller, a CFI may be very cost effective. For very large land areas (e.g., forests of Brazil, Canada, and Russia), with limited road access, transportation costs (e.g., helicopter access) would limit the CFI to a very low intensity of plots. This would preclude the possibility of obtaining accurate estimates for smaller subsets of the land area (i.e., scaling down in space). Also, delineating the spatial boundaries to determine the area associated with each stratum is challenging when plots are widely dispersed in space.

Where costs are very high, sampling with partial replacement has been used, although this can be quite difficult to analyze with more than two time measures (e.g., Roesch & Realms, 1999; Johnson et al., 2003). Using spatial modeling, including kriging approaches (e.g., universal kriging), estimates of carbon components at smaller spatial scales (e.g., soil carbon) can be obtained for any place in the inventory area (e.g., Mueller & Pierce, 2003). The use of spatial modeling increases the complexity of analyses resulting in more difficult explanations, and repeatability is not assured, as a model of the spatial variability must be selected, and this can be somewhat subjective. Spatial models will not be very accurate, when ground plots are widely dispersed on a highly diverse land area (e.g., mountainous terrain). Also, some biomass components, such as above-ground biomass, are often spatially clumped, and difficult to model using spatial models. The recent shift to continuous cover forestry and partial harvests has increased the challenge of monitoring even using ground plots, because of the increasing spatial variability (Iles & Smith, 2006).

3.2 "Top-Down" Approach

The "top-down" approach involves separating the forest land into smaller areas. Information is obtained for each spatial scale by coupling data from a variety of inventory data sources, with spatial or non-spatial modeling, including imputation and averaging, to estimate some or all of these carbon sources and sinks (e.g., Beets et al., 1999; Rohner & Böswald, 2001; Li et al., 2002; Wulder et al., 2003). For some approaches, the forest land is divided down to the stand (polygon) level (Fig. 3), whereas for other approaches, the smallest spatial scale is the pixel (Fig. 4).

Costs of acquiring, processing, storing, and displaying the remotely sensed information are a large part of project costs. For very large forested areas, such as those of Russia, Canada, and Brazil, this in itself is a challenge as a large number of remotely sensed images is needed. Ground data are often used to build and validate models that can be linked to remotely sensed data. In application, often ground data needed to drive the model are much less than for a "bottom-up" approach and imagery can be used to detect larger changes to the land area at reasonable costs (e.g., Franklin et al., 2002).

In the past, the extent of forest land versus other land uses was based on aerial photographs at smaller scales (e.g., 1:50,000 for coarse separation of forest types), whereas lower resolution satellite imagery (e.g., Landsat) is now sometimes used for this purpose. Separating the land into forest land versus other land uses, such as agriculture, is often not possible using only low resolution satellite imagery. For example, young forests can often have similar reflectances to agricultural crops

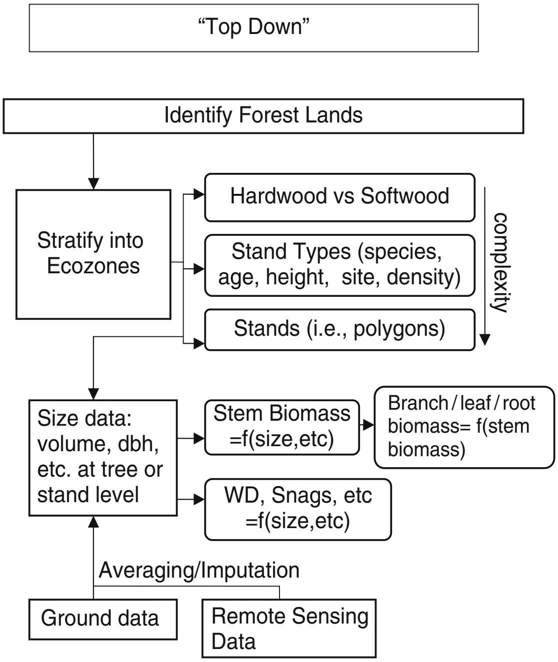


Fig. 3 "Top-down" approach using remotely sensed data to identify forest lands, and to stratify the forest land, coupled with ground and other remote sensing detail to obtain information for each stratum, down to the stand level of complexity

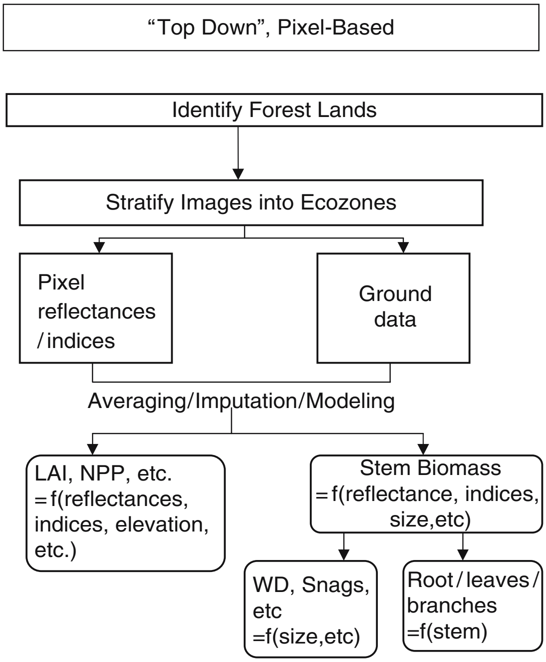


Fig. 4 Variation on the "Top-down" approach with a greater emphasis on remotely sensed data to obtain information down to the pixel level

(e.g., Suratman et al., 2004). Also, recent changes to continuous cover forestry, with more partial harvests replacing clearcutting, have made these forest areas more difficult to identify using remotely sensed imagery, since the areas do not correspond well with pixel boundaries of satellite imagery, there is high within pixel variability, and boundaries are harder to determine even on large scale (e.g., 1:2,000) aerial photographs or other high resolution imagery (e.g., IKONOS data with 1 m resolution). If forest land can be adequately delineated on remotely sensed images, actual changes in forest land can be easily detected using repeated images. Identifying the reasons for these changes (e.g., fires, land slides, harvest, urban conversion, etc.) may require high-level resolution imagery, including aerial photographs, coupled with a mixture of ground information, including land-use surveys.

A variety of approaches have been used to estimate carbon stocks using remotely sensed imagery based on the approach illustrated in Fig. 3. Heikkinen et al. (2004) used ground measures of carbon dioxide and methane fluxes in different vegetation types, and coupled these with classification using Landsat TM[2] imagery to obtain

---

[2] Thematic mapper.

estimates for East European tundra. In other applications, remotely sensed data were coupled with a carbon budget model. For carbon budget models that require stand summary information as model inputs (e.g., Li et al., 2002), a number of research studies have indicated that satellite imagery can be used to separate deciduous, coniferous, and mixed deciduous/coniferous forests. Other carbon models require detailed forest type information including sizes (heights, etc.), ages, and density (or cover) that is difficult to obtain using satellite imagery (e.g., Suratman et al., 2004). In the past, this detailed forest information was obtained via labor-intensive photo-interpretation of larger scale aerial photographs (e.g., 1:15,000) to segregate into stand types. Mechanization of this segregation may be achieved via very high resolution imagery such as LiDAR[3] (e.g., Lim & Treitz, 2004) data, but this is just now being researched, may be too expensive to implement, and would require extremely large databases for storage, retrieval, and analysis. Neeff et al. (2005) used SAR[4] data to obtain stand structure detail and estimated basal area and above-ground biomass in tropical forests. For other variables, such as ages in complex stands and the amount and type of coarse woody debris, supplemental ground data are needed. Multivariate approaches such as variable-space nearest neighbour methods (i.e., imputation) can be used to obtain estimates at local scales (i.e., scaling down), by using remotely sensed data to impute within stand details for sampled to non-sampled stands (e.g., Maltamo & Kangas, 1998; Moeur, 2000; McRoberts et al., 2002; Tomppo et al., 2002; Temesgen et al., 2003). However, a very low ground sampling intensity is unlikely to produce good estimates at this smaller scale (e.g., Katila, 2004; LeMay & Temesgen, 2005a). Moreover, matching ground data to strata (stands or pixels) is often difficult (Halme & Tomppo, 2001; LeMay & Temesgen, 2005b).

Using the general approach outlined in Fig. 4, pixel reflectances and derived indices could be used to obtain estimates of net primary productivity and greenhouse gases for above-ground carbon stocks (e.g., Coops & Waring, 2001). Imputation approaches that link ground-measured variables to pixels may be used to estimate non-sampled areas (e.g., Tomppo et al., 2002). Ground-measured variables can also be used to help correct for errors in map data from remotely sensed imagery (Katila et al., 2000). This information typically does not provide insights into below-canopy carbon stocks in dead organic matter, litter, soil carbon or below-ground biomass. This type of approach is already being used for the greenhouse gas accounting system in Australia (Richards, 2002).

Determining the accuracy of “top-down” approaches that rely on a mixture of models with remotely sensed and ground data is not simple, or sometimes, not possible. Assessments of individual model or inventory components do not always indicate the overall accuracy, since errors in one model or level in the inventory process may “cancel out” errors in another model or level (Kauppi, 2003) or error confounding can occur (Gertner, 2003). Validation methods, often using data

---

[3] Light detection and ranging.

[4] Airborne interferometric X and P-band synthetic aperture radar.

splitting techniques, are used to check model components, for the original or new populations. If CFI data are available for a particular variable (e.g., measured stem biomass), these data can be used to validate the model estimates at the land area level, and for sub-areas. A complication occurs if the data used to develop the model are also used to test the model, since this test over-estimates model accuracy. Cost often prohibits the availability of independent data for model validation. For carbon models, this is more problematic since, as noted, obtaining measures of some carbon pools is destructive and very expensive.

3.3 Models Used in Both Approaches

For the "top-down" approach, models that estimate carbon stocks are often developed for particular species and land areas, and then adopted for use in other populations. Simularly, for the "bottom-up" approach, some components may be estimated from other variables rather then measured (e.g., root biomass estimated from tree diameter). To improve local estimates, mixed-modeling approaches to impute to local areas are becoming more common (e.g., Robinson & Wykoff, 2004), along with other imputation approaches (e.g., Katila, 2004). Also, as management practices change to more spatially diverse stands, some models may not provide accurate estimates of stand dynamics, and therefore, carbon budgets. Models with process components are often used to help alleviate these problems, rather than strictly relying on empirical prediction equations (e.g., Beets et al., 1999).

4 Integrating Multiple Data Sources Across Time Scales

Using the CFI approach, the carbon budget can be reported for all of the times represented in the CFI. Change estimates are obtained by subtracting values from the previous time period. However, often this simple subtraction does not result in "real" estimates of change since:

1. Definitions of monitored elements have changed. For example, the definition of what is forest land and administrative boundaries may have changed.
2. Measuring devices may improve in precision over time.
3. Often, some measures are not available ("missing data") and must be estimated.
4. For variables that are not directly measured, improvements to equations may have resulted in different estimates. For example, equations to estimate biomass from tree diameter may have improved with further biomass sampling.

Some of these issues can be removed by re-analysis of data from both time periods using the same equations, definitions of forest land, etc. However, there is no way to correct for improvements in measurement precision and other differences.

For the "top-down" approach, the remotely sensed data used to stratify the landscape to the desired level of complexity may not be for the same date as other data sources used to obtain within strata detail. For areas that have little change, such as slow growing forests away from human settlements, pooling data sources within + or −5 years may be sufficient. For other areas with more frequent changes, failure to synchronize dates of various sources may have greater impacts on the uncertainty associated with the estimates.

Regardless of whether the "top-down" or "bottom-up" approaches are principally used to estimate carbon for periods included in the inventory, projections to future times (i.e., scaling up in time), and for future differences resulting from climate change, and/or future forest management activities are needed (e.g., Cao & Woodward, 1998; Beets et al., 1999; Kurz & Apps, 1999). Changes in climate can be incorporated into the models, as many carbon budget models include process model components. Also, since carbon models are inherently designed to estimate across scales via aggregation of smaller scale and/or separation of larger scale estimates, consistency across scales may be well represented (Mäkelä, 2003). Estimates of the impacts of natural and human caused disturbances may also be obtained from the models, although many of these impacts are currently being researched.

5 Concluding Remarks and Research Needs

Forests can be terrestrial sinks or sources for atmospheric carbon. Changes in forest management practices may increase the ability of forests to act as sinks, offsetting some of the emissions created through burning of fossil fuels. A variety of approaches has been used to estimate forest carbon stocks and changes by administrative unit, and to scale to larger or smaller units. Because of the need to forecast under changes in climate and management activities and to scale down to increasingly smaller spatial scales (small-area estimation) with more variables, the general trend appears to be towards more complex modelling approaches that include:

1. Greater use of remote sensing data as these data become available, particularly for land-use change monitoring, and integration of these with other data sources. This results in growing challenges in database management, storage, and analysis, however, particularly if very high spatial resolution imagery (e.g., IKONOS, coupled with LiDAR data) is used. Also, integration of data sources obtained at different times and for different spatial scales can be problematic.
2. Increased use of more recently developed modeling and estimation methods. These methods include: (1) mixed-models to estimate across scales and obtain logical consistency of estimates for aggregated (e.g., hardwood/softwood) versus segregated (e.g., species) spatial scales; (2) imputation and/or spatial modeling approaches to obtain information in non-sampled areas, to be used as model inputs, and/or in estimating carbon, and (3) systems of equations fitted simultaneously to obtain logical consistency and efficiency.

3. Greater use of models with process-components, to model possible responses to changes in climate and management activities, and to forecast future carbon stock changes.

However, ground measurements of carbon and carbon processes are needed to develop, calibrate, and validate the models used. This is particularly true since calculating accuracy for models coupled with remotely sensed data is extremely difficult. Confidence in the estimates must therefore come from validation of model components and overall estimates, whenever possible.

In terms of monitoring, research on methods for gathering, storing, and processing data for this and other forest inventory projects is needed. In addition, research specifically for carbon dioxide and greenhouse gases is needed. A partial listing of research questions includes:

1. Can high resolution imagery be used to replace photo-interpretation to obtain greater within strata detail (e.g., IKONOS, coupled with LiDAR data and Landsat TM data)?
2. How can multiple sources of information be integrated into an overall imputation approach to improve accuracy with minimal increases in cost, including computational issues and methods issues?
3. How can mixed-effects and systems of equations be best used in estimating between time scales, and to scale down to lower (more detailed) levels, while maintaining consistency across components and across scales?
4. What are the impacts of forest management and natural disturbances on greenhouse gas emissions and sinks, including dead organic matter and soil carbon, and how can this information be used for small and large-area estimation?

Improvements to forest inventory systems, including more detailed ground measurements of dead wood, soil carbon, and other attributes related to greenhouse gas emissions and sequestration, will improve our ability to make decisions regarding changes in forest management practices that may increase net carbon sinks in our forests.

References

Akindele, S.O., LeMay, V.M. (2006). Development of tree volume equations for common timber species in tropical rain forest areas of Nigeria. *Forest Ecology and Management, 226*, 41–48.

Beets, P.N., Robertson, K.A., Ford-Robertson, J.B., Gordon, J., MacLaren, J.P. (1999). Description and validation of C_change: a model for simulating carbon content in managed *Pinus radiata* stands. *New Zealand Journal of Forestry Science, 19*(3), 409–427.

Bi, H., Turner, J., Lambert, M.J. (2004). Additive biomass equations for native eucalypt forest trees of temperate Australia. *Trees, 18*, 467–479.

Binkley, C.S., Brand, D., Harkin, Z., Bull, G., Ravindranath, N.H., Obersteiner, M., Nilsson, S., Yamagata, Y., Krott, M. (2002). Carbon sink by the forest sector—options and needs for implementation. *Forest Policy and Economics, 4*, 65–77.

Cairns, M.A., Brown, S., Helmer, E.H., Baumgardner, G.A. (1997). Root biomass allocation in the world's upland forests. *Oecologia, 111*(1), 1–11.

Cao, M.K., Woodward, F.I. (1998). Dynamic responses of terrestrial ecosystem carbon cycling to global climate change. *Nature, 393*, 249–252.

Chave, J., Andalo, C., Brown, S., Cairns, M.A., Chambers, J.Q., Eamus, D., Fölster, H., Fromard, F., Higuchi, N., Kira, T., Lescure, J.-P., Nelson, B.W., Ogawa, H., Puig, H., Riéra, B., Yamakura, T. (2005). Tree allometry and improved estimation of carbon stocks and balance in tropical forests. *Oecologia, 145*, 87–99.

Christensen, J.H., Hewitson, B., Busuioc, A., Chen, A., Gao, X., Held, I., Jones, R., Kolli, R.K., Kwon, W.-T., Laprise, R., Magaña Rueda, V., Mearns, L., Menéndez, C.G., Räisänen, J., Rinke, A., Sarr, A., Whetton, P. (2007). Regional climate projections. In Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.). *Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambrige: Cambridge University Press.

Cochran, W.G. (1977). *Sampling techniques*, 3rd ed. Toronto: Wiley.

Coops, N.C., Waring, R.H. (2001). The use of multi-scale remote sensing imagery to derive regional estimates of forest growth capacity using 3-PGS. *Remote Sensing of Environment, 75*(3), 324–334.

Fang, J., Chen, A., Peng, C., Zhao, S., Ci, L. (2001). Changes in forest biomass carbon storage in China between 1949 and 1998. *Science,* 292, 2320–2322.

Franklin, S.E., Lavigne, M.B., Wulder, M.A., Stenhouse, G.B. (2002). Change detection and landscape structure mapping using remote sensing. *Forestry Chronicle, 78*(5), 618–625.

Ganjegunte, G.K., Condron, L.M., Clinton, P.W., Davis, M.R., Mahieu, N. (2004). Decomposition and nutrient release from radiata pine (*Pinus radiata*) coarse woody debris. *Forest Ecology and Management, 187*, 197–211.

Gertner, G. (2003). Assessment of computationally intensive spatial statistical methods for generating inputs for spatially explicit error budgets. *Forest Biometry, Modelling and Information Sciences, 1*, 27–34. Retrieved March 2006, http://www.fbmis.info

Gower, S.T., McMurtrie, R.E., Murty, D. (1996). Aboveground net primary production decline with stand age: potential causes. *Trends Ecological Evolutions, 11*(9), 378–382.

Halme, M., Tomppo, E. (2001). Improving the accuracy of multisource forest inventory estimates by reducing plot location error – a multicriteria approach. *Remote Sensing of Environment, 78*(3), 321–327.

Heikkinen, J.E.P., Virtanen, T., Huttunen, J.T., Elsakov, V., Martikainen, P.J. (2004). Carbon balance in East European tundra. *Global Biogeochemical Cycles, 18*, GB1023, 14 p.

Iles, K., Smith, N. (2006). A new type of sample plot that is particularly useful for sampling small clusters of objects. *Forest Science, 52*(2), 148–154.

IPCC. (2007). Summary for policymakers. In Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.). *Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge: Cambridge University Press.

Jenkins, J.C., Chojnacky, D.C., Heath, L.S., Birdsey, R.A. (2003). National-scale biomass equations for United States tree species. *Forest Science, 49*(1), 12–35.

Johnson, D.S., Williams, M.S., Czaplewski, R.L. (2003). Comparison of estimators for rolling samples using forest inventory and analysis data. *Forest Science, 49*(1), 50–63.

Katila, M. (2004). *Controlling the estimation errors in the Finnish multisource National Forest Inventory*. Research Papers (dissertation) 910. Finish Forest Research Institute.

Katila, M., Heikkinen, J., Tomppo, E. (2000). Calibration of small-area estimates for map errors in mulitsource forest inventory. *Canadian Journal of Forest Research, 30*, 1329–1339.

Kauppi, P.E. (2003). New, low estimate for carbon stock in global forest vegetation based on inventory data. *Silva Fennica, 37*(4), 451–457.

Kurz, W.A., Apps, M.J. (1999). A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. *Ecological Applications, 9*(1), 526–547.

Kurz, W.A., Beukema, S.J., Apps, M.J. (1998). Carbon budget implications of the transition from natural to managed disturbance regimes in forest landscapes. *Mitigation and Adaptation Strategies for Global Change, 2*, 405–421.

Kurz, W.A., Apps, M., Banfield, E., Stinson, G. (2002). Forest carbon accounting at the operational scale. *Forestry Chronicle, 78*(5), 672–679.

Lambert, M.-C., Ung, C.-H., Raulier, F. (2005). Canadian national tree aboveground biomass. *Canadian Journal of Forest Research, 35*, 1996–2018.

LeMay, V., Temesgen, H. (2005a). Comparison of nearest neighbor methods for estimating basal area and stems per ha using aerial auxiliary variables. *Forest Science, 51*(2), 109–119.

LeMay, V., Temesgen, H. (2005b). Connecting inventory information sources for landscape level analyses. *Forest Biometry, Modelling, and Information Sciences,* 1, 37–49. Retrieved March 2006, http://www.fbmis.info.

Li, Z., Apps, M.J., Banfield, E., Kurz, W.A. (2002). Estimating net primary production of forests in the Canadian Prairie Provinces using an inventory-based carbon budget model. *Canadian Journal of Forest Research, 32*(1), 161–169.

Li, Z., Kurz, W.A., Apps, M.J., Beukema, S.J. (2003a). Belowground biomass dynamics in the Carbon Budget Model of the Canadian Forest Sector: recent improvements and implications of the estimation of NPP and NEP. *Canadian Journal of Forest Research, 33*, 126–136.

Li, Z., Apps, M.J., Kurz, W.A., Banfield, E. (2003b). Temporal changes of forest net primary production and net ecosystem production in west central Canada associated with natural and anthropogenic disturbances. *Canadian Journal of Forest Research, 33*, 2340–2351.

Lim, K.S., Treitz, P.M. (2004). Estimation of above ground forest biomass from Airborne Discrete Return Laser scanner data using canopy-based quantile estimators. *Scandinavian Journal of Forest Research, 29*, 558–570.

Luken, J.O., Hinton, A.C., Baker, D.G. (1991). Forest edges associated with power-line corridors and implications for corridor siting. *Landscape and Urban Planning, 20*(4), 315–324.

Mäkelä, A. (2003). Process-based modelling of tree and stand growth : towards a hierarchical treatment of multiscale processes. *Canadian Journal of Forest Research, 33*(3), 398–409.

Maltamo, M., Kangas, A. (1998). Methods based on k-nearest neighbor regression in the prediction of basal area diameter distribution. *Canadian Journal of Forest Research, 28*, 1107–1115.

McRoberts, R.E., Nelson, M.D., Wendt, D.G. (2002). Stratified estimation of forest area using satellite imagery, inventory data, and the k-Nearest Neighbors technique. *Remote Sensing of Environment, 82*, 457–468.

Moeur, M. (2000). Extending stand exam data with most similar neighbor inference. *Proceedings of the Society of American Foresters National Convention, September 11–15, 1999.* Portland, OR, pp. 99–107.

Mouillet, F., Field, C.B. (2005). Fire history and the global carbon budget: a 1° × 1° fire history reconstruction for the 20th century. *Global Change Biology, 11*, 398–420.

Mueller, T.G., Pierce, F.J. (2003). Soil carbon maps: enhancing spatial estimates with simple terrain attributes at multiple scales. *Soil Science American Journal, 67*, 258–267.

Myneni, R.B., Dong, J., Tucker, C.J., Kaufmann, R.K., Kauppi, P.E., Liski, J., Zhou, L., Alexeyev, V. (2001). A large carbon sink in the woody biomass of northern forests. *Proceedings of the National Academy of Science, 98*(26), 14784–14789.

Nabuurs, G.J., Masera, O., Andrasko, K., Benitez-Ponce, P., Boer, R., Dutschke, M., Elsiddig, E., Ford-Robertson, J., Frumhoff, P., Karjalainen, T., Krankina, O., Kurz, W., Matsumoto, M., Oyhantcabal, W., Ravindranath, N.H., Sanz Sanchez, M.J., Zhang, X. (2007). Forestry. In Metz, B., O.R. Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (eds.) *Climate change 2007: Mitigation. Contribution of Working group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change,* Cambridge, UK: Cambridge University Press.

Neeff, T., Dutra, L.V., dos Santos, J.R., da Costa Freitas, C., Araujo, L.S. (2005). Tropical forest measurement by interferometric height modeling and P-band radar backscatter. *Forest Science, 51*(6), 585–594.

Nelson, J. (2003). Forest-level models and challenges for their successful application. *Canadian Journal of Forest Research, 33*(3), 422–429.

Nuutinen, T., Kellomäki, S. (2001). A comparison of three modelling approaches for large-scale forest scenario analysis in Finland. *Silva Fennica, 35*(3), 299–308.

Oliver, G.R., Beets, P.N., Garrett, L.G., Pearce, S.H., Kimberly, M.O., Ford-Robertson, J.B., Robertson, K.A. (2004). Variation in soil carbon in pine plantations and implications for monitoring soil carbon stocks in relation to land-use change and forest site management in New Zealand. *Forest Ecology and Management, 203*, 283–295.

Penman, J., Gytarsky, M., Hiraishi, T., Kurg, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F. (eds.) (2003). *Good practice guidance for land use, land-use change and forestry*. Japan: Intergovernmental Panel on Climate Change (IPCC/IGES).

Richards, G.P. (2002). *The FULLCAM carbon accounting model: Development, calibration and implementation for the National Carbon Accounting System*. NCAS Technical Report No. 28. Canberra: Australian Greenhouse Office.

Robinson, A., Wykoff, W.R. (2004). Imputing missing height measure using a mixed-effects modeling strategy. *Canadian Journal of Forest Research, 34*, 2492–2500.

Roesch, F.A., Realms, G.A. (1999). Analytical alternatives for an annual inventory system. *Journal of Forestry, 97*, 33–37.

Rohner, M., Böswald. K. (2001). Forestry development scenarios: timber production, carbon dynamics in tree biomass and forest values in Germany. *Silva Fennica, 35*(3), 277–297.

Schimel, D., Melillo, J., Tian, H., McGuire, A.D., Kicklighter, D., Kittel, T., Rosenbloom, N., Running, S., Thornton, P., Ojima, D., Parton, W., Kelly, R., Sykes, M., Neilson, R., Rizzo, B. (2000). Contribution of increasing CO_2 and climate to carbon storage by ecosystems in the United States. *Science, 287*, 2004–2006.

Smith, J.E., Heath, L.S. (2002). *A model of forest floor carbon mass for United States forest types*. Research Paper NE-722. USDA Forest Service.

Suratman, M.N., Bull, G.Q., Leckie, D.G., LeMay, V., Marshall, P.L., Mispan, M.R. (2004). Prediction models for estimating volume, age, predicting area of rubber (*Hevea brasiliensis*) plantations in Malaysia using Landsat TM data. *International Forestry Review, 6*(1), 1–12.

Temesgen, H., LeMay, V.M., Froese, K.L., Marshall, P.L. (2003). Imputing tree-lists from aerial attributes for complex stands of south-eastern British Columbia. *Forest Ecology and Management, 177*, 277–285.

Tomppo, E., Nilsson, M., Rosengren, M., Aalto, P., Kennedy, P. (2002). Simultaneous use of Landsat-TM and IRS-1C WiFS data in estimating large area tree stem volume and aboveground biomass. *Remote Sensing of Environment, 82*(1), 156–177.

Vågen, T.G., Lal, R., Singh, B.R. (2005). Soil carbon sequestration in Sub-Saharan Africa: A Review. *Land Degradation and Development, 16*, 53–71.

White, T.M., Kurz, W.A. (2005). Afforestation on private land in Canada from 1990 to 2002 estimated from historical records. *Forestry Chronicle, 81*(4), 491–497.

Wulder, M.A., Kurz, W.A., Gillis, M. (2003). National level forest monitoring and modeling in Canada. *Progress in Planning, 61*, 365–381.

Yang, Y.S., Guo, J., Chen, G., Xie, J., Gao, R., Li, Z., Jin, Z. (2005). Carbon and nitrogen pools in Chinese fir and evergreen broadleaved forests and changes associated with felling and burning in mid-subtropical China. *Forest Ecology and Management, 216*, 216–226.

Zeide, B. (2003). The U-approach to forest modeling. *Canadian Journal of Forest Research, 33*(3), 480–489.

Forest Eco-physiological Models and Carbon Sequestration

T. Keenan[1]*, S. Sabaté[2], and C. Gracia[3]

1 Introduction

Modeling and monitoring the processes involved in terrestrial carbon sequestration are often thought to be independent events. In fact, rigorously validated modern modeling techniques are very useful tools in the monitoring of the carbon sequestration potential of an ecosystem through simulation, by highlighting key areas for study of what is a complex dynamical system. This is ever more important in the light of climate change, where it becomes essential to have an understanding of the future role of terrestrial ecosystems as potential sinks or sources in the global carbon cycle.

The study of the effects of climate change on terrestrial ecosystems is one field of interest which requires the use of predictive tools such as functional simulation models. There are many possible applications of such models, from studying the responses of individual processes, the interactions of various processes, up to the responses of whole forest stands and ecosystems. This can be performed focusing on the response of forests to climate change (and in turn identifying feedbacks from forest ecosystem responses that may affect the rate of climate change), the effect of climate change on ecosystem service supplies which are necessary for societies well being (such as water supply, soil fertility and productivity), the effect of management on forest productivity, or in assessing the suitability of a certain site for plantation.

Models can be taken as quantitative predictors of ecosystem responses by translating a particular "stress" of interest to a key ecosystem parameter, taking into account a margin of error, but perhaps more importantly, they give us a way to scale up our understanding of individual process reactions to drivers on the individual tree level to the ecosystem scale. The quantitative predictions have a large range of uncertainty, and are actually by no means predictions, but estimates. However, the

---

[1] Ph.D. candidate with CREAF, Autonomous University of Barcelona, Barcelona, Spain
Phone number +34935812915 * t.keenan@creaf.uab.es

[2] Researcher with CREAF, and a Professor with the Department of Ecology, University of Barcelona, Barcelona, Spain

[3] Researcher with CREAF, and a Professor with the Department of Ecology, University of Barcelona, Barcelona, Spain

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

qualitative descriptions of ecosystem responses give a very valuable insight into the functioning and potential response of the ecosystem as a whole.

In this chapter, we first outline the different approaches to forest eco-physiological modeling, with their associated pros and cons, and applications. We then give an example of the application of one such forest growth model, Gotilwa+, to sites along a latitudinal transect in Europe, as an example of how the method can be applied throughout Europe. The methodology is then extended to the application of the model to the whole of Europe for the coming 100 years, with an exploration of the forest eco-physiological responses to climate change, in particular the effects on carbon and water balances.

1.1 Forest Services

Modelling can prove a useful tool in assessing the expected future state of forest ecosystem services (e.g. water availability, soil fertility, wood production, fire hazard reduction etc.) that are vital for human well being. This is of particular interest in the light of climate change. Global change is continually altering such services, and is expected to do so to an even greater extent in the future. Previous Europe wide studies have applied terrestrial ecosystem models such as those described in this chapter to asses the expected future status of services such as soil fertility, water availability and the risk of forest fires (Schröter et al., 2005). Both positive and negative trends were reported, with increases of forest area and productivity on one hand, but an increase in the risk of fire, and a decrease in soil fertility and water availability on the other.

By applying the assumed changes in land use and climate, the models can be used to gauge the effect of such changes on ecosystem services. The Gotilwa+ model presented in this chapter has been involved in such studies and uses the same approach to assessing the future of European forests and ecosystem service supply. This is often coupled with an assessment of possible management strategies to assess the capability of forest management to offset or counteract any potentially negative or undesired effects.

1.2 Applications in Forest Management

Modelling can also be used to assess the potential of forest management strategies. Forest management practices aim to optimise the productivity of the forest and minimise the risk posed by environmental stresses. The suitability of a management strategy is highly dependant on site characteristics and the general state of the forest stand. It is a difficult balance to achieve, where over-harvesting can lead to serious damage to a forest ecosystem, whilst under-harvesting can fail to make full use of the ecosystems potential, or indeed lead to a significant loss of aboveground biomass (e.g. in the case of fire). Models allow for the evaluation of many alternative

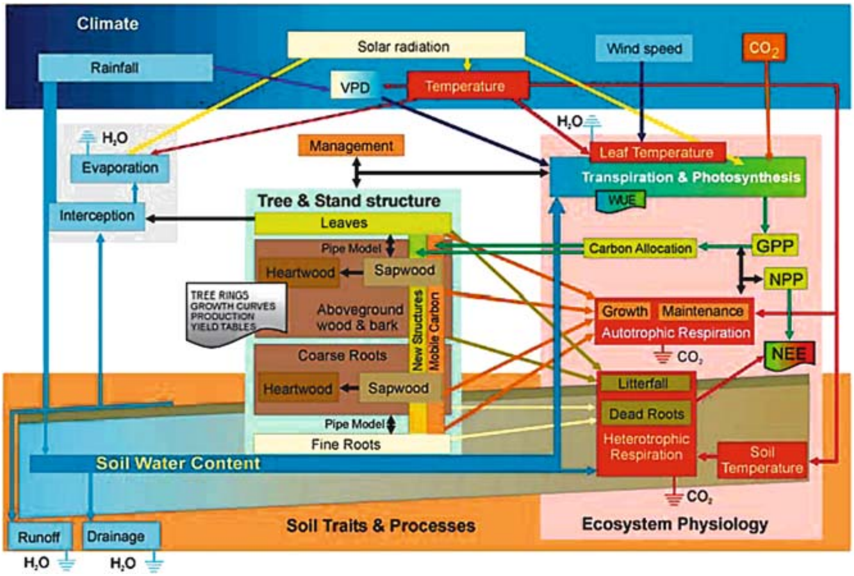


Fig. 1 A schematic graph of processes and interactions accounted for in the GOTILWA+ model (Kramer et al., 2002)

strategies, and the effectiveness of each can thus be tested based on the requirements of the manager. This is relevant both in the maximising of the potential for the ecosystem to sequester carbon, and in protecting ecosystems which are threatened by changing environmental conditions (with the aim to be to give the system more time to adapt naturally, and avoid threshold limits) (Kellomaki and Valmari, 2005).

This 'virtual management' allows the forest manager to enter the forest and invoke management strategies, with the potential to remove selected trees based on different removal criteria, either at prescribed intervals based on a certain value such as average diameter at breast height, or at regular time steps. The value of this virtual management is that it immediately gives the forest planner the results of his strategy for the future. The effect of the strategy can be focused on maximising whatever variable the planner is interested in, or indeed finding the optimal maximum considering a variety of requirements. It is important to note that today's management strategy for a particular site might not be suitable in a changing world, and a modelling approach testing a range of plausible strategies can warn a planner of the need of a strategy change before damage is done to the ecosystem.

1.3 Process Based Models Versus Empirical Models

There are two main approaches available to modellers: The empirical approach and the process based approach. The choice of approach taken is highly dependent on the problem being addressed. As always, both approaches have valid applications, each with their own strengths and weaknesses. In reality, the two options are not quite independent, with many models containing a synergy of the two approaches.

Empirical models attempt to simplify the system description, by relying purely on known system wide responses to external drivers. They are statistically based, are easy to feed (require less parameters) and generally have faster execution times.

This is very useful, making it easy to build a simple and accurate description of a system with very few parameters. As their name suggests, they are based on empirical functions, which attempt to describe direct ecosystem responses. This simplicity and speed also helps in the analysis of model results, and is useful in giving insight into the general functioning of a system, highlighting the key processes and possible reactions. However the applicability of empirical models is restricted and their application as true exploratory tools is questionable. Limited by their simplicity and their basis of empirical responses, they lack the ability to explore new scenarios and conditions outside of those on which they were built and tested.

Process based models, in contrast, are complex simulators that attempt to mimic the real world. The aim is to include mathematical descriptions of both the processes that govern a system, and their interactions, thus recreating the system in a virtual environment. Each process in the system is described separately, and dynamically interacts with other processes. Given an accurate description of each processes separately, it is argued that a better description of the ecosystem in general, through the interaction of these processes, can be achieved. Due to their detail, a large number of parameters are necessary. The parameters determine the response of each function describing an individual process, and are based on detailed field work or lab experiments. This allows an accurate description of all factors affecting a process, but such parameters are not always available. This can be a problem, and a lack of data often leads to assumptions and approximations, but the approach leads to a model with a wide applicability. The detail and dynamic characteristic of process based models allows them, theoretically, to function as effective exploratory tools and they should be fully applicable under new conditions and scenarios.

The scientific community is often somewhat sceptical about the effectiveness of complex process based simulation models, and the role they should play in ecological studies. Many ecologists will laugh if you explain that you are trying to mimic the real world. And indeed they might! The environment is highly variable, and could be said to be the most complex system in existence. However, complex process based models can have a much wider applicability than that of simpler empirical models that are simply designed to fit data. Although far more complicated than empirical models, and much more expensive to build, they give an insight into the internal functioning of the system itself, which could never be achieved with empirical models. For studies involving climate change, this is essential, as complex process are involved in ecosystem wide responses to global change. Unfortunately, our current understanding of many processes is still too limited to allow fully process based modelling, and most so called processed based models use a range of semi to fully empirical equations. This is perfectly valid, but one must keep in mind that most process based models, including the Gotilwa+ model, are actually hybrids of the two approaches.

2 GOTILWA+: A Process-Based Model

2.1 The Model

GOTILWA+ (Growth of Trees Is Limited by WAter) (Gracia et al., 1999), is a process based forest growth model that has been implemented to simulate the processes underlying growth and to explore how these processes are influenced by climate, tree stand structure, management techniques, soil properties and climate change. The Gotilwa+ model simulates carbon and water fluxes through forests in different environments, for different tree species, under changing environmental conditions, either due to climate or to management regimes.

Results of GOTILWA+ are computed separately for 50 DBH (diameter at breast height) classes and they are integrated at the stand level. The processes are described with different sub-models that interact and integrate the results of simulated growth and the evolution of the whole tree stand through time (hourly calculations integrated at a daily time step). Horizontal space is assumed homogeneous and the vertical profile distinguishes two canopy layers (sun and shade conditions).

2.1.1 Input and Output Variables

The input data includes: climate (max. and min. temperatures, rainfall, VPD, wind speed, global radiation and atmospheric CO_2 concentration); stand characteristics (tree structure including the structure of the canopy; DBH class distribution); tree physiology (photosynthetic and stomatal conductance parameters, specific growth and maintenance respiration rates), site conditions including soil characteristics and hydrological parameters and also forest management criteria.

Many output variables can be extracted from the model. These can be separated into three main categories: canopy variables, tree and stand structural variables, and root and soil variables.

Canopy variables include: Gross Primary Production, Net Primary Production, Net Ecosystem Exchange, Leaf Area Index, Transpiration, Water Use Efficiency, Leaf Production, Leaf Respiration, Leaf Biomass, Growth Activity, The Length of the Growing Period and Volatile Organic Compound emissions.

Tree and Stand structural variables include: Tree Density, Sapling Density, Basal Area, Sapwood Area, Mean Quadratic Tree Diameter, Vigour Index, Tree Height, Wood Production, Wood Respiration, Mobile Carbohydrates, Tree Ring Width, Aboveground Biomass, the Weight of the Sapwood Column, Wood Volume, Dead Wood Volume and Yield (when considering management).

Root and Soil variables include: Soil Temperature, Water Stored in Soil, Fine and Gross Litter Fall, Soil Organic Carbon, Fine Root Biomass, Fine Root Production, Fine Root Respiration, Heterotrophic Respiration, Maintenance Respiration and Growth Respiration.

2.1.2 How GOTILWA+ Copes with Processes

Process based models start at the very basic physiological leaf level, combining and describing the different processes involved. Figure 2 shows a schematic of the most fundamental compartment, the leaf. Here, photosynthesis is calculated dynamically, based on internal and external conditions.

Gotilwa+ comprises of a two layer canopy photosynthetic model, coupled with a carbon allocation and growth model and a soil respiration and hydrology model. It describes monospecific stands, which can be even or uneven aged. The key environmental forcing factors taken into account are precipitation, air temperature, vapour pressure, global radiation, wind speed, and atmospheric carbon dioxide concentration. Using this data, the response of ecosystem processes is calculated to estimate the carbon and water fluxes in a forest ecosystem. It is an individual based model, where individual trees in the forest are grouped into 50 DBH (Diameter at Breast Height) classes, with calculations being performed separately for each class. Stand characteristics are taken into account and species specific parameters, to give highly accurate predictions of forest growth and carbon or water fluxes in the system.

The two layer canopy photosynthesis submodel splits the available leaf area index into sun and shade leaves, depending on the time of the day, leaf area angle

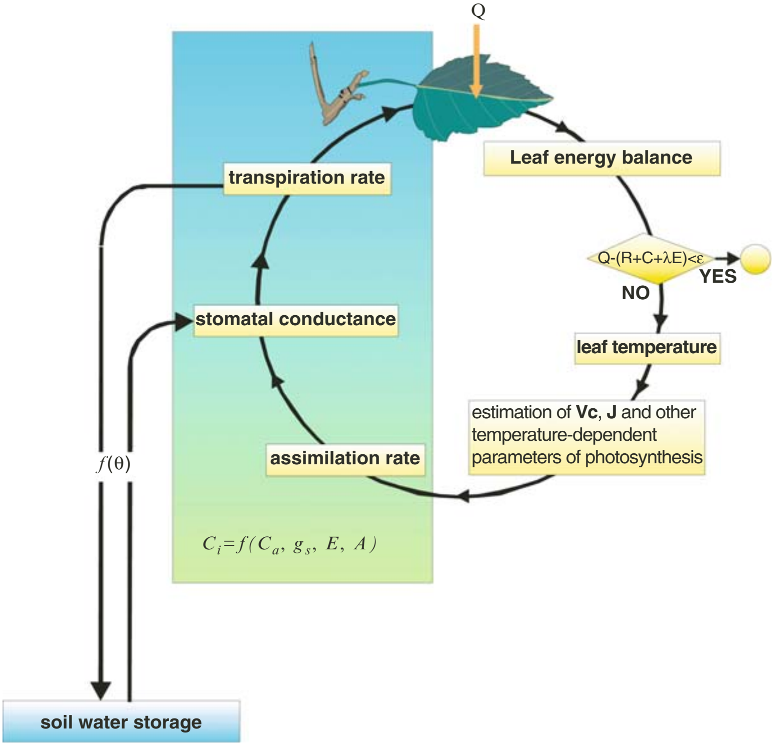


Fig. 2 A schematic diagram of the representation in GOTILWA+ of the photosynthetic assimilation rate

and the canopy's ellipsoidal distribution. Assimilation rates depend on the direct and diffuse radiation intercepted, the species specific photosynthetic capacities, leaf temperature, available carbon, the extent of stomatal opening, and the availability of soil water.

Growth and the allocation of mobile carbon for tree maintenance are considered through three compartments: Leaf respiration, Sapwood respiration, and Fine roots respiration.

Fine litter fall (e.g. leaves), gross litter fall (e.g. bark, branches) and the mortality of fine roots add to the soil organic carbon content. The soil in Gotilwa+ is divided into two layers, an organic layer and a mineral layer, with a rate of transfer between them. Soil organic carbon is decomposed depending on to which layer it belongs, with both decomposition rates depending on a Q_{10} function taking into account soil water content and soil temperature. Soil temperature is calculated from air temperature using a moving average of 11 days. The amount of soil water available for organic layers is calculated taking into account the cumulated rainfall of the previous 30 days and soil water availability for mineral layers depends on the soils water filled porosity which in turn is a function of the organic matter present in soil.

Soil water content is described as one layer, taking inputs though precipitation less leaf interception, which is evaporated, (stem interception, or stemflow, is not evaporated, but directed to the soil), and outputs though drainage, runoff, and transpiration. Surface evaporation only occurs when the canopy is not closed.

Flux calculations are performed hourly, whereas slower processes such as growth and other state variables are calculated daily.

2.1.3 Model Validation

Gotilwa+ model validation has been carried out at various sites across Europe and the United States (Kramer et al., 2002; Morales et al., 2005; Keenan et al., 2008), using canopy level measurements gathered by the FLUXNET network. Figure 3 shows the results at one such site, a *Quercus ilex* forest in the south of France, giving 2 years of field data (2002, 2003) for daily Gross Primary Production, against modelled values. It can be seen that the model successfully captures both the high productivity in spring and autumn and the decline in summer due the effect of drought on the photosynthetic apparatus.

2.1.4 Unknowns in Forest Modelling

A correct description of each process is crucial. This requires intense and extensive field work, data collection and experimentation. Thus, by using field work to better our understanding of the processes involved and the factors that affect them, we can build more accurate models. There is yet a lot to be understood, and many interactions between species, soil and atmospheric processes are still poorly understood. Such factors include the role of belowground biomass (the "hidden half" of the forest), the effect of nutrient availability, factors affecting soil organic matter decomposition,

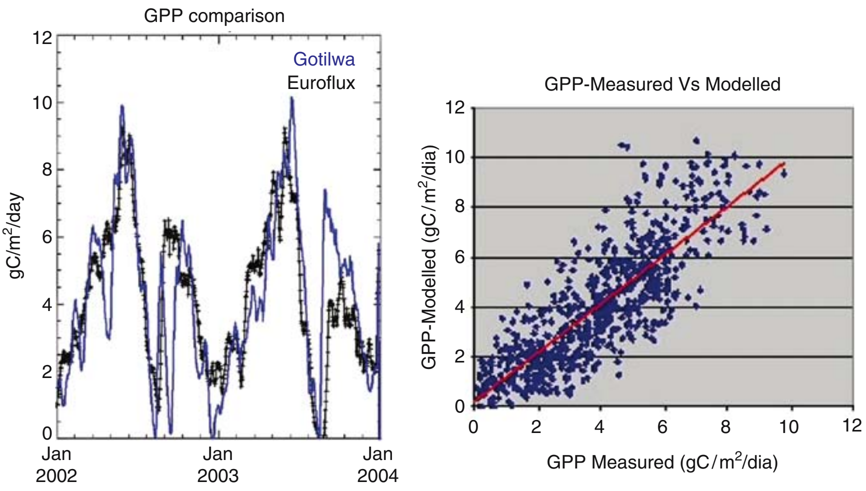


Fig. 3 Daily primary production (10 day smoothed) modelled by Gotilwa+, compared against field data gathered by the FLUXNET network at the Puechabon site, a *Quercus ilex* forest in southern France

and species specific responses to climate change factors such as elevated CO_2, drought and the role of acclimation.

This lack of information is exacerbated by the problem of scale. Many questions remain as to how processes scale up from the chloroplast or mitochondrial level, to the leaf, the stand, and the ecosystem as a whole. The problem of physiological scale is coupled by a problem of temporal scale. An ecosystem incorporates many processes, each with their own temporal scale. Fast processes (such as the leaf energy balance, photosynthesis, stomatal conductance, transpiration, autotrophic and heterotrophic respiration, water and light canopy interception) interact with slow processes (tree ring formation, sapwood to heartwood changes, tree mortality, wood increment, management, soil decomposition, climate change). These questions are all approached with as much accuracy as possible in the model, but many factors could be improved. Such problems go hand in hand with any modelling attempt but each year we are improving our knowledge, and our ability to use it.

3 Model Applications

3.1 The Future of European Forests – A Case Study on a Latitudinal Gradient

Before presenting the results from Europe-wide simulations, we give, as an example, results from simulations at select sites along a latitudinal gradient. Theoretically, any tree species can be simulated by the Gotilwa+ model, given the availability of species

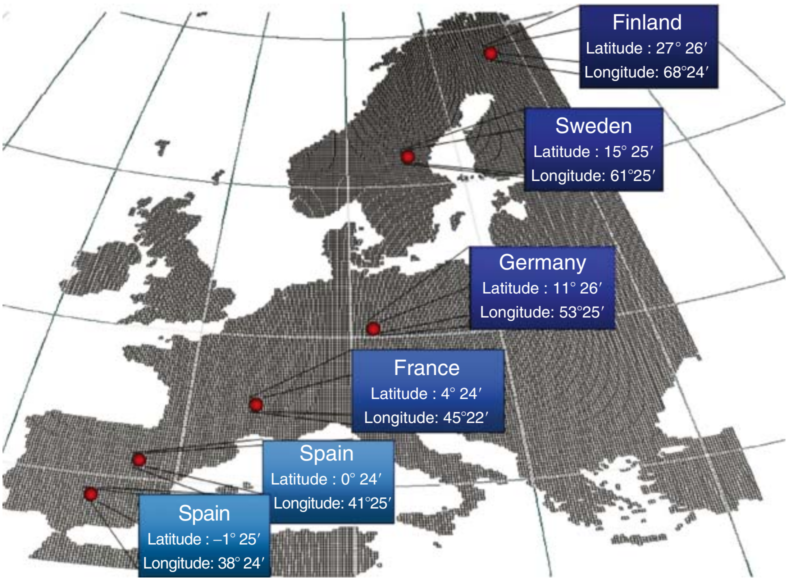


Fig. 4 A map of selected pixels with Pinus sylvestris stands in Europe

Table 1 Pixel climate details – current yearly averages

Country	Lon	Lat	Q	P	dP	PET	Min T	Max T	VP
1 Finland	27°26′	68°24′	1,991	448	194	513	–6.41	2.10	0.51
2 Sweden	15°25′	61°25′	2,671	600	180	730	–2.36	7.12	0.61
3 Germany	11°26′	53°25′	3,226	607	225	805	5.19	12.11	0.98
4 France	4°24′	45°22′	4,270	847	191	944	4.09	13.07	0.89
5 Spain	0°24′41°25′		5,354	366	62	1,340	9.39	20.55	1.27
6 Spain	–1°25′38°24′		5,375	386	61	1,380	8.86	20.83	1.22

Column headings are: Lon – longitude, Lat – latitude, Q – global radiation ($MJ/m^2/year$), P – annual precipitation (mm/year), PET – potential evapotranspiration (mm/year), Min T/Max T – minimum/maximum temperature (^0C), VP – vapour pressure (kPa) dP – number of days with precipitation

specific parameters. Here, we report on the modelling of *Pinus sylvestris*, as the predominant species along the gradient. Each site in Fig. 4 is the site of a current *Pinus sylvestris* stand. Site locations and climatic conditions are given in Table 1.

3.1.1 Future Climate Data – A Multi GCM Ensemble

Past climate data is often locally available, and the Climate Research Unit at Norwich has developed an extensive database of reconstructed climate (CRU) for the past 100 years (Mitchell et al., 2004). This is used to ‘spin up’ the model, and validate the key processes.

Projecting forest growth into the future is highly dependant on the climate data used to run the model. The best tools available for predicting future climate evolution are Global Climate Models or General Circulation Models (GCMs). GCMs aim to describe climate behavior by integrating a variety of fluid-dynamical, chemical, or even biological equations that are either derived directly from physical laws (e.g. Newton's Law) or constructed by more empirical means. A large number of GCMs exist for predicting future climate evolution. Each applies the laws of physics and mathematical descriptions of atmospheric interactions to varying degrees to give a prediction for the evolution of future climate.

A range of socio-economic scenarios has been developed to explore future paths of carbon emissions related to the burning of fossil fuels. These can be used to force GCMs. This approach is currently used by the IPCC (Inter-governmental Panel on Climate Change) is used as a driver for the GCMs, giving various possible future greenhouse gas emissions, depending on the economic model applied and the resulting changes in population, land use change and energy consumption. Four emissions scenarios are derived from the IPCC's SRES1 (The global Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios): A1 Fossil-Intensive, A2, B1, and B2, ranging from pessimistic to optimistic regarding future anthropogenic impact on the climate system.

A large difference exists between the predictions of each of the GCMs, and each of the scenarios. They differ in: (a) their climate sensitivity and (b) the spatial pattern of change, making multi model assessments essential for a good understanding of potential changes.

Here a multi model ensemble was used to provide a probability distribution function for per pixel climate evolution. Thus, inter-model and inter scenario uncertainty can be assessed, and the effect of this uncertainty on terrestrial ecophysiological models can be gauged. Data from four GCMs were used, with each one qualified through its use by the IPCC Data Distribution Centre. The specific data used was compiled through the ATEAM (Advanced Terrestrial Ecosystem.

Analysis and Modelling, www.pik-potsdam.de/ateam) project, and ALARM project (Assessing Large-scale Risks for biodiversity with tested Methods, www.alarmproject.net/alarm).

Climate data from the following four GCMs were applied along the latitudinal gradient:

1. The HadCM3 model from the Hadley Centre in England (Mitchell et al., 1998)
2. The NCAR-PCM model from the National Centre of Atmospheric Research, USA, which has the smallest sensitivity of all models compared to forcing at the global scale (Washington et al., 2000)
3. The CSIRO2 model from the Commonwealth Scientific and Industrial Research Organisation, Australia, which has above-average climate sensitivity, a little higher than HadCM3 (Flato & Boer, 2001)
4. The CGCM2 model from the Canadian Centre for Climate Modelling and Analysis (Laprise et al., 2003)

Results from the multi model ensemble for the latitudinal gradient give the mean predicted values for the range of ecosystem indicators available, but also give a measure of the uncertainty associated from the choice of climate model and scenario, as can be seen in Fig. 5.

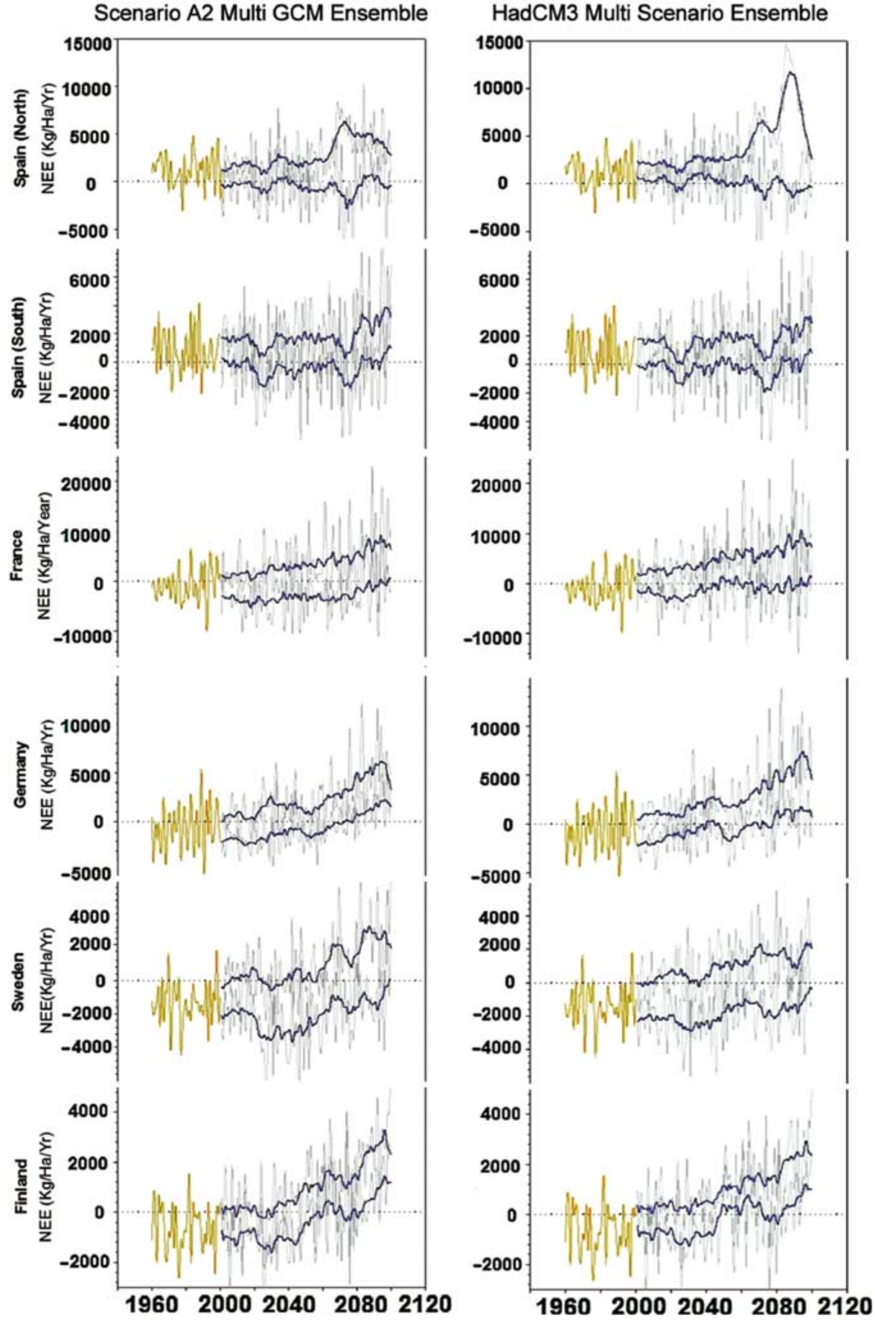


Fig. 5 **Left**: The evolution of GOTILWA+ outputs of NEE at each site, with CRU climatology until 2000 and an ensemble of GCMs using the A2 climate scenario, from 2000 to 2001. **Right**: The evolution of NEE at each site, with CRU climatology until 2000, and an ensemble of climate change scenarios with the HadCM3 GCM from 2000 to 2100. The grey lines represent the maximum and minimum range of NEE, and the blue line gives the 10 year running average of the maximum and minimum range

Although the climate models and scenarios vary in their predictions, they agree in qualitative terms and there is a general consensus that, although it would refine the results, increased accuracy would not change the conclusion with regards to many ecosystem variables. From Fig. 5, the effect of climate change predicted by the models and scenarios can be seen. Here, each ecosystem is predicted to become a net source of carbon, thus constituting a positive feedback on the climate system. The amount of uncertainty in this prediction is broken down into two categories: the uncertainty that derives from the choice of emissions scenario, and the uncertainty that derives from the choice of GCM to supply the climatic variables.

As can be seen from Fig. 5, the choice of which GCM to use accounts for almost as much variability as the choice of socio-economic scenario. That the variation due to the choice of emission scenario used is only slightly greater than that associated with the choice of GCM, highlights that important climate processes are imperfectly accounted for by these climate models.

3.1.2 Stand Performance at the Selected Sites

Given the computational cost of performing simulations with an entire multi model ensemble, most studies choose a particular model and scenario. Here, to ease the interpretation of the results, we present results from the HadCM3 model predictions with the A2 emission scenario as our description of future climate (this gives mid-range levels of future climate change). Figure 6 shows the Gross Primary Production and Net Primary Production predicted by the model for each pixel of the transect using the HadCM3 models climatic variables. Here an increase in GPP can be observed, resulting from higher temperatures and CO_2 fertilisation. This trend is followed by all sites, except the more northern Spanish site, which suffers high mortality. This increase in GPP leads to only a slight increase in NPP, as higher production is balanced by higher respiration rates.

A wide variety of variables are available as output from the model. Here we present those of a potentially greater interest, such as Wood Production and Aboveground Biomass as shown in Fig. 7 below.

Wood Production and Above Ground Biomass are strongly coupled variables, and show a strong response to the imposed climate change conditions. Both show a marked increase at sites where water stress is not a restricting factor.

3.2 The Effect of Management

Management can play a very important role in ecosystem function. In the model, various different management regimes and strategies can be defined, and their effect on the forest ecosystem can be gauged. Such strategies are often focussed on optimizing carbon sequestration, wood production, yield, or aboveground biomass. Particular interest in the Mediterranean region is focused on using management to

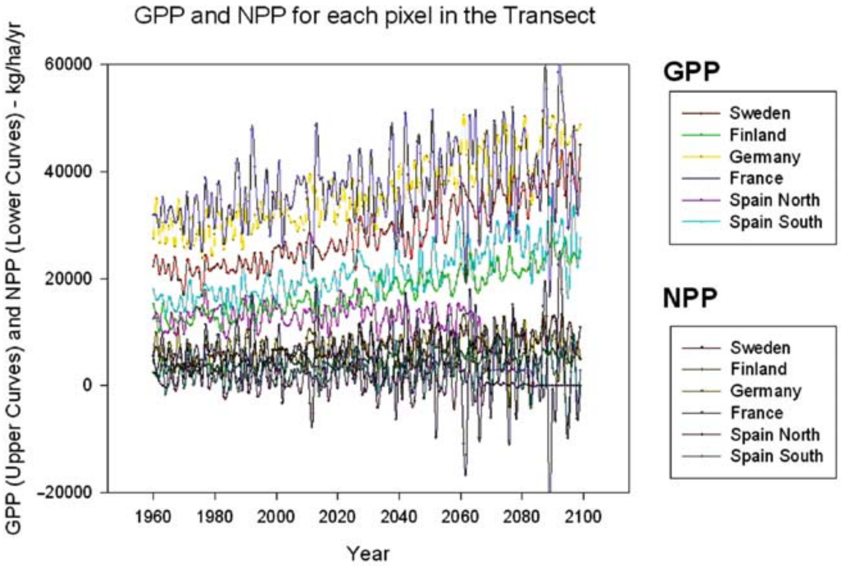


Fig. 6 The GOTILWA+ projection of gross primary production and net primary production at each site, between 1960 and 2100, using HadCM3 – A2

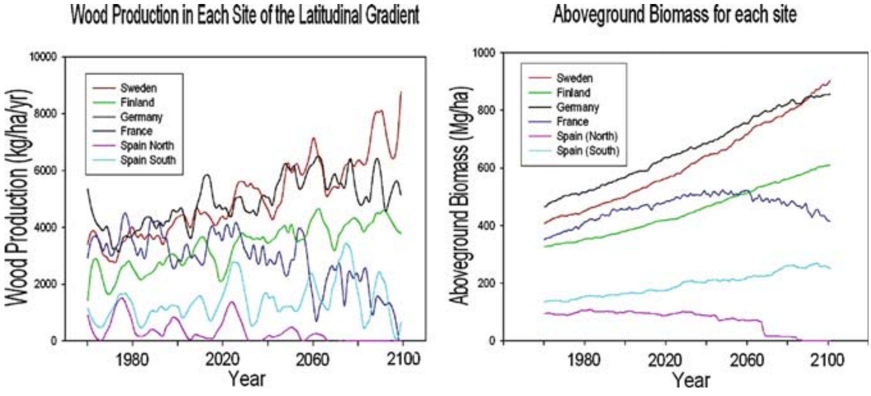


Fig. 7 The GOTILWA+ projection of wood production (**Left**) and aboveground biomass (**Right**) at each site, between 1960 and 2100

mitigate the effects of drought on forest stands. Here, an example is given of a management strategy applied to the forest stand at the French site. The management strategy in this simulation is to enter the forest when the basal area reaches 42 m^2/ha and remove the larger trees until a basal area is reached of 38 m^2/ha. This has the effect of increasing wood production, while giving a high yield from the system, thus increasing the capacity of the stand to act as a net sink for atmospheric

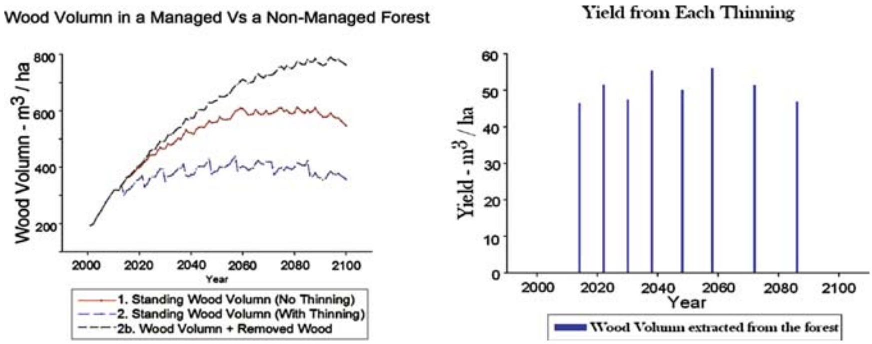


Fig. 8 **Left**: The GOTILWA+ projection of wood volume remaining in an unmanaged forest and in the same forest with management. **Right**: The wood volume extracted from the managed forest at each intervention in the simulation

CO_2. This strategy can be contrasted against alternatives, and an optimum strategy found, depending on the prerequisites of the user.

Figure 8 shows how management can increase the productivity of the forest, with the total wood volume (Yield + Standing Volume) at the end of 100 years being greater in the managed forest than in the unmanaged forest. This occurs due to the response of the forest to decreased competition for resources. It has been argued to increase the lifetime of a forests sequestration capacity. It can also increase the capacity of the forest to act as a sink of atmospheric CO_2. On the other hand this also depends on what use the extracted wood is put to. The mean life time of wood products is estimated to be about 30 years, though this is highly dependant on the product, thus any additional sink that results from the extraction of wood from the system can be presumed to be short lived.

3.3 The Future of European Forests – Europe-wide Simulations

Gotilwa+ has been validated extensively in Europe (Kramer et al., 2002; Morales et al., 2005). This has helped refine the model, and now the same modelling approach can be applied throughout Europe. This can be a very useful tool for those monitoring future carbon sequestration trends in European forests. To supply the input data required by the model, an extensive database has been built within the framework of the European ALARM project (Assessing Large-scale Risks for biodiversity with tested Methods, www.alarmproject.net/alarm), connecting diverse information sources at a European level and adapting them to fit the same spatial resolution.

The database contains data related to forest functional types, forest cover, forest structure (tree density and size distribution), forest function (photosynthesis, respiration rates), soil hydrology, organic matter decomposition rates and management strategies. This data base provides the model with all the necessary information to run in each pixel and it also provides the climatic series at this level of detail for different climate change scenarios generated by several general circulation models (GCMs). Given the computational expense of running GOTILWA+ with the predictions of each GCM and each climate scenario, we chose the HadCM3 GCM with the IPCC scenario A2 to simulate future forest stands over Europe.

Here, in Fig. 9, we see a shift in the majority of European forest ecosystems from being net sinks of carbon to net sources of carbon. This reflects what we observed earlier for the latitudinal transect in Fig. 5. It represents a potential feedback on the climate system, where terrestrial ecosystems themselves do not help to solve the problem of climate change and may even serve to augment it. Currently, most are acting as sinks, effectively removing and storing carbon from the atmosphere.

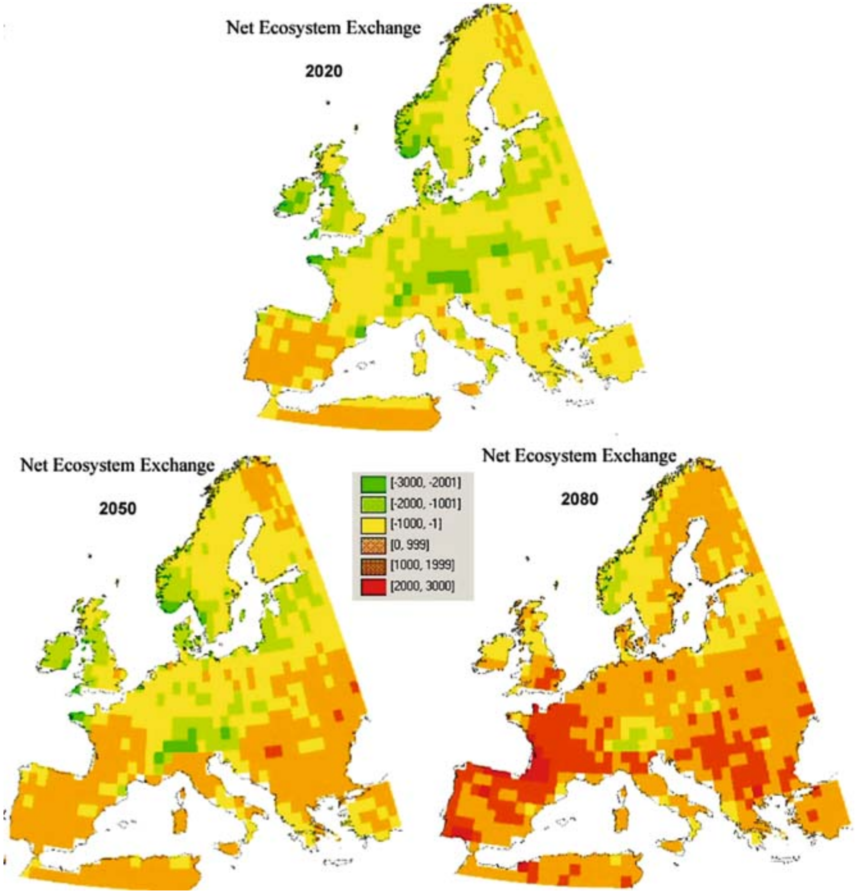


Fig. 9 GOTILWA+ projections of net ecosystem exchange (kg/ha/year) from Pinus sylvestris forests in Europe wide simulations (values represent the annual average for each time slice 2020: 2010 to 2030, 2050: 2040 to 2060, and 2080: 2070 to 2090)

The perspective of them becoming sources is not a pleasant thought, with vast amounts of carbon currently stored in soils, and ready for release.

Figure 10 allows us to further explore this response. As can be seen, productivity, in general (in areas not under stressed conditions) is expected to increase, thus constituting an increase in the ability of the ecosystem to remove carbon from the atmosphere. The conversion of the ecosystem to a net source of carbon results from the reaction of respiration rates and the large available pool of carbon in the soil. The description of soil respiration is as good as our current understanding of these processes allows (Fang et al., 2005; Jannsans et al., 2005), though undoubtedly further work is required to reduce our uncertainty. Figure 5 gives an estimate of the uncertainty which is attributable to the uncertainty in predictions of climatic data, but the same conclusion can be applied here, that the quantitative conclusion may be questionable, but the qualitative conclusion should not vary greatly with a better understanding of the processes involved.

In Fig. 11, a general tendency for a decrease in soil water content can be observed. This is due more to higher Evapotranspiration rates, from a combination of increased productivity and higher temperatures, than to changes in the distribution of precipitation. The effects are expected to be more extreme in the Mediterranean region, where soil water content is already extremely low. This can have repercussions outside of the ecosystems in question, effecting other ecosystems and society at large.

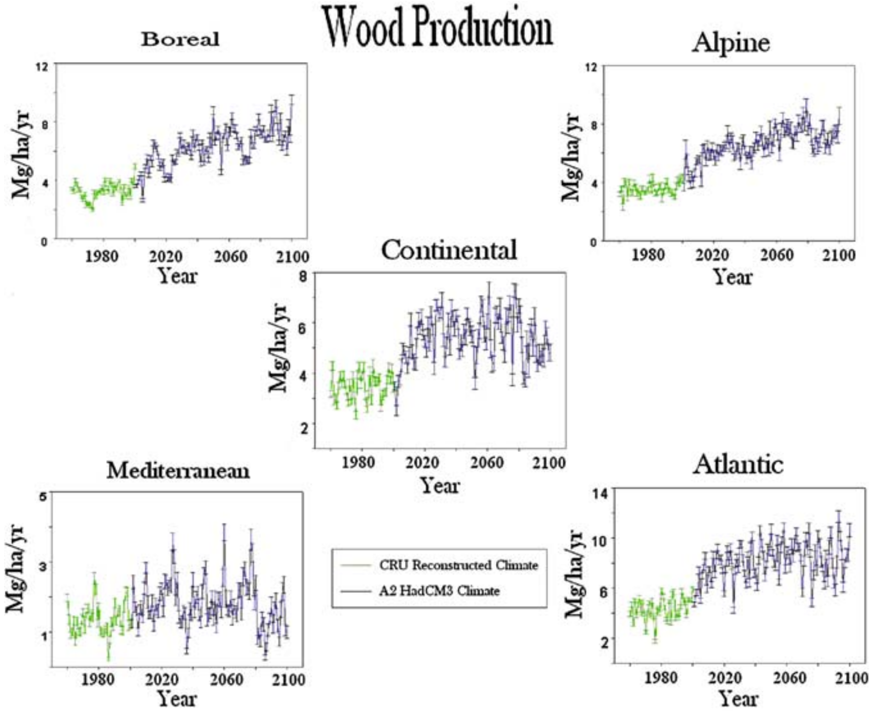


Fig. 10 GOTILWA+ projections of wood production (Mg/ha/year) for five main European climate regions from 1960 to 2100

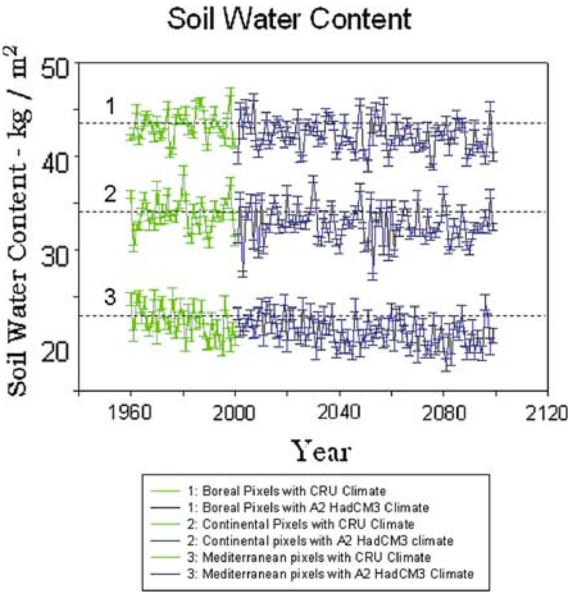


Fig. 11 GOTILWA+ projections of the quantity of water in the soil for three climate types, Boreal, Continental, and Mediterranean climates, with reconstructed climate from 1960 to 2000, and the Hadley Center HadCM3 model with scenario A2 until 2100

4 The Future of Eco-physiological Models

As our knowledge of processes and ecosystem responses develops, in parallel with computing science, so too does our ability to build eco-physiological models with higher accuracy and a broader applicability. Future efforts will be focused both on the development of our scientific knowledge of the processes involved, and in using the models themselves to better our understanding of how these processes link together to form an ecosystem. This will be carried out through extensive field work and experimentation, from field trials to stand simulations, up to the coupling of vegetation ecosystem models with global climate models.

4.1 Climate Models and Eco-physiological Models

It has long been accepted that regional climate affects the local distribution of vegetation and soils, with natural undisturbed vegetation effectively mirroring the long term local climate (Koppen, 1936). In recent years, a broader understanding of the interaction between vegetation and climate has been developed. Not only does climate effect the distribution and functioning of vegetation, but vegetation also has an affect on climate, and the two are inextricably linked. This feedback mechanism is now recognised as being crucial to the evolution of the Earths climate (Bonan, 2002), and equally crucial in predicting the anticipated change in the earths climate in the future (Cox, 2000). Potentially one of the most interesting future prospects for eco-physiological models is their coupling with regional climate models, in an attempt to incorporate the dynamic relationship between vegetation and climate.

4.2 Development

Our current understanding of terrestrial processes is limited in many areas, with various key features only relatively weakly represented. The advancement of our understanding of these critical processes should better enable us to accurately model real world situations. This will be achieved by integrating the latest understanding in climatic, hydrologic and edaphic controls on forest ecosystem process, obtained from the analysis of intensive field and laboratory data, into novel model parameterisations.

The list is long, but key areas currently being developed include: the representation of soil organic matter decomposition, which is very variable and not always best described by a simple temperature-water relationship; the coupled Nitrogen cycle, which is being greatly altered throughout the world due to anthropogenic global change, and is at present very poorly understood; eco-physiological responses to elevated concentrations of atmospheric CO_2, and the problem of acclimatation; accurate descriptions of the functioning of belowground biomass, the hidden half of terrestrial ecosystems. Belowground biomass can account for half of the total biomass of a terrestrial ecosystem in the Mediterranean, but it is difficult to study; species interactions (competition/mutualism) provide one of the key problems in describing succession and dynamic vegetation problems; the role of Volatile Organic Compounds, which play a part in protection and the processing of assimilated carbon in many species, and Fire events.

5 Conclusion

Process based forest eco-physiological models are very useful tools and have a wide application through many streams of research. Their functions range from assessment tools for forest managers and policy makers, to predictive tools for studies on ecosystem functioning, to essential components of large scale global models of climate evolution. The concept of this chapter has been to give a general overview of the structure and applications of such models, using the process based model GOTILWA+ as an example. We have discussed both empirical models, and process based models, and their relative pros and cons, and used GOTILWA+ as an example of how their application can give useful insights into current and future ecosystem functioning, both on a local, regional, and indeed global scale.

Although our knowledge is far from complete, and qualitative results are associated with a large amount of uncertainty, it is a rapidly developing area of research, and state of the art techniques are constantly being applied to improve our understanding, and the ability to produce accurate results. Current efforts are focusing on using highly accurate field data (such as that produced by the EUROFLUX network, using eddy covariance techniques (www-eosdis.ornl.gov/FLUXNET)) to further validate the models over a wide range of site conditions and ecosystem structures. This newly available high quality data also allows us to highlight important processes that are not sufficiently described.

Little by little, as our understanding grows, so too does the capability of such models to accurately replicate real life processes. Here we have given an overview of the current state of the art of biogeochemical terrestrial modelling, although unfortunately, what has been presented is already out of date.

Acknowledgments This study was funded by the European Commission via a Marie Curie Excellence Grant through GREENCYCLES, the Marie-Curie Biogeochemistry and Climate Change Research and Training Network (MRTN-CT-2004–512464) supported by the European Commissions Sixth Framework program for Earth System Science. Data was supplied by the ALARM project (Assessing LArge-scale environmental Risks for biodiversity with tested Methods, GOCE-CT-2003–506675), from the EU Fifth Framework for Energy, environment and sustainable development. Invaluable assistance was also provided by Eduard Pla, and Jordi Vayreda.

References

Bonan, G. (2002). Ecological Climatology – Concepts and Applications. Cambridge University Press, Cambridge.

Cox, P. M. (2000). Acceleration of global warming due to carbon cycle feedbacks in a coupled climate model. *Nature, 408*, 184–187.

Fang, C., Smith, P., Moncrieff, J., Smith, J. (2005). Similar responses of labile and resistant organic matter pools to changes in temperature. *Nature, 433*, 57–58.

Flato, G. M., Boer G. J. (2001). Warming asymmetry in climate change simulations. *Geophysical Research Letters, 28*, 195–198.

Gracia, C. A., Tello, E., Sabate, S., Bellot, J. (1999). GOTILWA+: An integrated model of water dynamics and forest growth. ecology of Mediterranean evergreen oak forests. In: Rodà, F., Retana, J., Bellot, J., Gracia, C. A. (eds.) Ecology of Mediterranean Evergreen Oak Forests. Springer, Berlin, pp. 163–178.

Jaansans, I. A., Freibauer, A., Schlamadinger, B., Ceulemans, R., Ciais, P., Dolman, A. J., Heimann, M., Nabuurs, G. J., Smith, P., Valentini, R., Schultz, E. D. (2005). The carbon budget of terrestrial ecosystems at the country scale – a European case study. *Biogeosciences, 2*, 15–26.

Keenan, T., García, R., Zaehle, S., Friend, A., Gracia, C., Sabate, S. (2008). An Analysis of Ecophysiological Responses to Drought in the Mediterranean: Data Analysis and a Comparison and Evaluation of Process Based Ecosystem Model Predictions in Drought Prone Environments (In review).

Kellomaki, T., Valmari, A. (2005). Method for analysing the performance of certain testing techniques for concurrent systems. *Fifth International conference on the Application of Concurrency to System Design, Proceedings*, pp. 154–163.

Koppen, W. (1936). Das geographische System der Klimate. In: Koppen, W., Geiger, R. (eds.) Handbuch der Klimatologie, 5, Gedbruder Borntrager, pp. 110–152.

Kramer, K., Leinonen, I., Bartelink, H. H., Berbigier, P., Borghetti, M., Bernhofer, C. H., Cienciala, E., Dolman, A. J., Froer, O., Gracia, C. A., Granier, A., Grunwald, T., Hari, P., Jans, W., Kellomaki, S., Loustau, D., Magnani, F., Matteucci, G., Mohren, G. M. J., Moors, E., Nissinen, A., Peltola, H., Sabate, S., Sanchez, A., Sontag, M., Valentini, R., Vesala, T. (2002). Evaluation of 6 process-based forest growth models using eddy-covariance measurements of CO^2 and H_2O fluxes at six forest sites in Europe. *Global Change Biology, 8*, 1–18.

Laprise, R., Caya, D., Friqon, A., Paquin, D. (2003). Current and perturbed climate as simulated by the second-generation Canadian Regional Climate Model (CRCM-II) over northwestern North America. *Climate Dynamics, 21* (5–6), 405–421.

Mitchell, J., Johns, T., Senior, C. (1998). Transient response to increasing greenhouse gases using models with and without flux ajustments. Hadley Centre Technical Note 2. UK Met Office, London Road, Braknell, UK.

Mitchell, T. D., Hulme, M. (1999). Predicting regional climate change: living with uncertainty. *Progress in Physical Geography, 23* (1), 57–78.

Mitchell, T. D., Carter, T. R., Jones, P. D., Hulme, M., New, M. (2004). A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed record (1901–2000) and 16 scenarios (2001–2100). Working Paper 55. Tyne Centre for Climate Change Research, University of East Anglia, Norwich (for CRU TS 2.02).

Morales, P., Sykes, M. T., Prentice, I. C., Smith, P., Smith, B., Bugmann, H., Zierl, B., Friedlingstein, P., Viovy, N., Sabaté, S., Sánchez, A., Pla, E., Gracia, C. A., Sitch, S., Arneth, A., Ogee, J. (2005). Comparing and evaluating process-based ecosystem model predictions of carbon and water fluxes in major European forest biomes. *Global Change Biology*, *11*, 2211–2233.

Schröter, D., Cramer, W., Leemans, R., Prentice, C., et al. (2005). Ecosystem service supply and vulnerability to global change in Europe. *Science, 310*, 1333–1337.

Washington, W. M., Weatherly, J. W., Meehl, G. A., Semtner, A. J., Bettge, Jr., T. W., Craig, A. P., Strand, W. G., Arblaster, Jr., J. M., Wayland, V. B., James, R., Zhang, Y. (2000). Parallel climate model (PCM) control and transient simulations. *Climate Dynamics, 16* (10/11), 755–774.

Influence of Climatic Variables on Crown Condition in Pine Forests of Northern Spain

A. V. Sanz-Ros*, J. A. Pajares, and J. J. Diez

1 Introduction

Climate Change over the last century has created concern to the scientific community, as it could have a major impact on natural and social systems at local, regional and national scales Current mitigation policies derived from Kyoto Protocol are following two main ways: reduction of gas emissions, and implementation of a sustainable development assuring persistence of greenhouse carbon sinks, mainly forests lands.

Sustainable forest management is an essential tool to assure the permanence of our forests and to maintain properly their ecological functioning. Some sustainable models proposed that terrestrial ecosystems together with bioenergy systems, including capturing and storing carbon, may even neutralize unsustainable historical carbon emissions in the course of a century (Obersteiner et al., 2001). They can offer a permanent carbon sink by the technological option of capturing carbon from biomass conversion facilities (Kraxner et al., 2003). But the role of forest as a CO2 sink could be influenced by the occurrence of forest pests and diseases, causing tree defoliation and canopy reduction. Climatic factors could influence crown condition, question that is tried to answer with this work.

Crown condition is closely related to forest condition, and also, the contribution of each individual tree to CO2 sequestration depends on its crown development. Visual assessment of defoliation became accepted as the standard method for large-scale intensive monitoring of forest condition in Europe, and it has been systematically assessed since 1986 throughout the whole Europe (EC & UN/ECE, 2000).

Estimation of crown density (or crown transparency), method, described by Innes (1990), and harmonized (Innes, 1993) using guidelines proposed by ICP Forests (1992), has been widely used as an indicator of the vitality of forest trees and of the degree of damage (Zierl, 2002). Canopy transparency (CT) concept is

---

Forest Entomology and Pathology Laboratory, Plant Production and Forest Resources Department, University of Valladolid, Avenida de Madrid 44, 34004 Palencia, Spain,
*tonisanz@pvs.uva.es, jdcasero@pvs.uva.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

wider than defoliation, because it takes into consideration factors like unusual reduction of leaf size, presence of flowers and cones, branching deformation or shoot death, and also premature needle loss (Ferretti, 1994), so that defoliation is included in this term. There are several causes of premature needle loss, sometimes they are well known (pests and diseases), but in many occasions they are far from clear, ranging from environmental stress (Zierl, 2004), such as low availability of water or extreme values of temperature, to other variables related to the management or disturbance events. Drought is a major factor in forest decline, making tree more vulnerable to fungi and pest attacks (Wellburn, 1994; Klap et al., 2000). In Mediterranean climate, growth of forest trees is subjected to many climatic constrains, particularly the availability of water (Gracia et al., 1999). It is known that some climatic factors can influence crown condition, but it is not known how this influence is, and which are these climatic parameters for each region. It is expected that variation of the climatic factors would be different among the diverse regions in future climate change scenarios, so its needed to consider climate trends obtained by several surveys at different scales.

Some studies indicate that rainfall would have a general decrease in south Europe to the Mediterranean (Schönwiese & Rapp, 1997; Piervitali et al., 1997; Buffoni et al., 1999; Brunetti et al., 2000, 2001). It seems that, in Spain, annual rainfall shows a trend towards a decrease over the whole Iberian Peninsula, the greatest decreases occurring in summer, but being the winters wetter (Karas, 1997; Esteban-Parra et al., 1998; Hulme & Sheard, 1999; Parry, 2000; IPCC, 2001; Mossman, 2002). In any case, some review showed an increased variability of precipitation everywhere (Dore, 2005). Temperature records show an increase in the global mean temperature between 0.4°C and 0.8°C along the 20th century that cannot be attributed to the internal variability of the climate system (Panel on Reconciling Temperature Observations, 2000; Parry, 2000). Other studies showed a global warming rate of 0.3– 0.6°C since the 19th century, due to either anthropogenic (IPCC, 2001) or to astronomic causes (Soon et al., 2000; Landscheidt, 2000). In addition, some studies pointed to that the last decade (1990–1999) was the warmest record, both annually and for the winter season.

This increase in the global temperature is not homogeneously distributed on the earth surface, varying among the different regions and locations. According to this, climate models currently have predicted a temperature increase at different scales. The Third Assessment Report projections for the present century, on a global scale, are that average temperature rise by 2100 would be in the range of 1.4–5.8°C (IPCC, 2001). Other models have forecasted approximately an increase of 1.5–3°C up to the year 2100 in Europe (Kattenberg et al., 1996), or between 1–3.5°C for mid-latitude regions (Watson et al., 1997). For the Iberian Peninsula, results seem to indicate an increase in the annual mean temperature of about 1.6°C over the last hundred years, with highest increases in summer (approximately 2°C) and the lowest in winter (Hulme & Sheard, 1999; Parry, 2000). This change is also reflected in the behaviour of the extreme values, which showed significant trends in some regions of the globe, but not in others, where no significant changes were detected (DeGaetano, 1996; Heino et al., 1999; Bonsal et al., 2001). Previous analysis from various surveys showed that the behaviour of extreme temperatures and their associated impact strongly depended on local conditions.

In some respects, these climate changes are likely to act as an important driving force on natural systems (Parmesan & Yohe, 2003). The increase of temperature along the next hundred years would be equivalent to a poleward shift of the present geographic isotherms of approximately 150–155 km (Watson et al., 1997), causing changes in forest tree species distributions. Risk of pests and diseases will be increased due to this limits displacement, so that many species will be placed in an stressing environment. In this way, tree vigour of these species could decrease, leading to canopy decline manifested in symptoms as defoliation and discolouration. The aim of this study was to find relationships between crown condition and some climatic parameters to identify which are those having a main influence on crown condition, and how this influence is shown in the tree (crown transparency), and to contribute to the understanding of how these parameters will affect under future climate change scenarios.

2 Materials and Methods

In this study, 68 National Forest Inventory (NFI) plots were sampled from July to mid September of 2005. All plots were placed in a pilot zone in Palencia province (northwest of Spain, Fig. 1), and were covered by three *Pinus* species (37 by *P. sylvestris*, 22 by *P. nigra*, and 9 by *P. pinaster*).

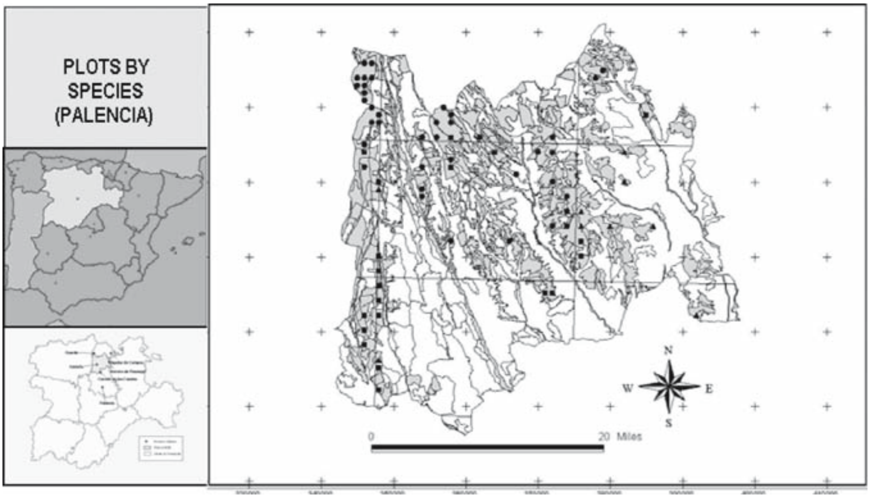


Fig. 1 Distribution of plots in pilot zone in Palencia Province, Castilla y León, Spain. Plots were taken from a 2 km grid on tree covered area. Pinus sylvestris (•), Pinus pinaster (▲), and P. nigra (■). Gray surface is forest covered area

Most of the plots were pine plantations, in some cases mixed with different oak and pine species. This area is transitional between agricultural lands (southwards) and Cantabric mountains (northwards), and extends for 186,642 ha, 60,000 of them forested, showing enough climatic variations to study the influence of climatic factors in crown condition. This pilot zone is located between UTM coordinates 342,000, 4,685,000, and 398,000, 4,741,000, ranging in altitude from 800 to 1,000 m a.s.l. (Fig. 1). The climate is Mediterranean with a slight Atlantic influence: 11.49°C of mean temperature and annual rainfall of 519 mm.

Sampling method involved four subplots (Fig. 2). One fixed subplot of 25 m radius (National Forest Inventory plot) and three subplot of 17.5 m radius, linked by two perpendicular linear transects of 50 m. In each subplot, the 20 nearest trees in a spiral pattern were evaluated. Data showed for canopy transparency for each plot were means of 20 evaluated trees The establishment of one subplot in a road or firewall, where an edge effect is likely, was avoided by subplot rotation. However some other surveys have showed that there is no differences in defoliation between inside stand trees and edge trees (Durrant & Boswell, 2002).

Canopy transparency was estimated in the field according to the European Programme for the Intensive Monitoring of Forest Ecosystems, Level I (ICP Forests, 1992). Crown density is a visual estimation of the amount of light passing through the tree crown relative to a reference tree with complete foliage. Canopy transparency is the opposite term, and is what was estimated in the present study comparing to reference pictures of canopy transparency for each species (Cadahia et al., 1991; Ferretti, 1994).

Climatic long-term data for each plot were obtained from the Digital Climatic Atlas of Iberian Peninsula (Ninyerola et al., 2005), a recent climatic model in which it is used data from all the meteorological stations from pilot zone, 15 of them within and other 31 in nearby areas. Rainfall values are refereed to the last twenty years means, and temperatures to the last fifteen years before publishing the model.

Several climatic variables were chosen (Table 1) to study their possible relation to crown condition, including annual temperature means and monthly values of dry and cold seasons, its rainfall and solar radiation. All of these climatic variables were categorized in five homogeneous intervals (Table 2) with the aim of comparing plot canopy transparency among different levels of each climatic variable.

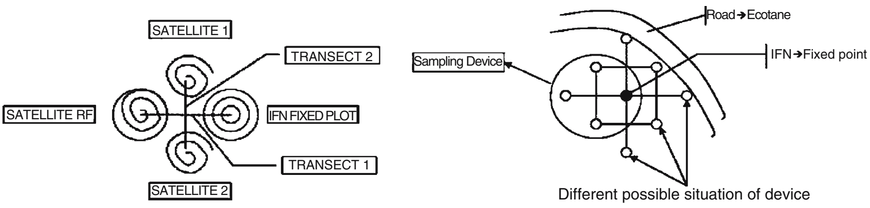


Fig. 2 Sampling method with four subplots and two linear transects linking them (Left). IFN plot is a fixed plot, original transect orientation was N-S and E-W, but it was able to be rotated in order to avoid roads or firewalls (Right)

Table 1 Likely predictor climatic variables used to find correlations with plot canopy transparency

	Annual	December	January	February	June	July	August
Mean temperature	x	x	x	x	x	x	x
Maximum temperature	x	x	x	x	x	x	x
Minimum temperature	x	x	x	x			
rainfall	x	x	x	x	x	x	x
Solar radiation	x						

Plot canopy transparency values were transformed by decimal logarithm to obtain normal distribution and homocedasticity of data (Kolmogorov-Smirnov, Shapiro-Wilks and Bartlett tests). Analysis of Variance (ANOVA), with a signification level of 0.05, was carried out to know if there were statistically significant differences in transparency values for the 68 sampled plots among the levels of rainfall, temperatures and solar radiation. Finally, the Bonferroni and Duncan test were used for multiple comparisons. To study the relationship between canopy transparency and climatic data, simple regression of untransformed data was used for each climatic variable, and multiple regression with backward selection was used with the aim of include several variables in the model to study cross effect among variables in crown transparency.

3 Results

The Kolmogorov-Smirnov, Shapiro-Wilks and Bartlett tests proved normality and homocedasticity of the logarithm of mean plot canopy transparency (CT) data. The One Way ANOVA analysis showed that there were statistically significant differences between mean CT values of plots with different levels of July rainfall, Mean annual temperature and mean August temperature, whereas there were no significant differences in plot CT in relation with the other variables analysed, such as solar radiation, mean annual or minimum temperatures, annual or winter rainfall (Table 3).

In the analysis of rainfall, ANOVA showed significant differences in plot CT among different homogeneous levels of July rainfall (Fig. 3, Top). Bonferroni test showed differences between levels 1 and 5 (Table 4), with a difference in precipitation of 60 mm. The erratic behaviour of rainfall distribution (Dore, 2005) could preclude from making accurate predictions for the future.

On the other hand, there were significant differences in plot CT among Mean annual temperature levels (Fig. 3, Medium). Duncan multiple comparison test demonstrated significant differences ($p < 0.05$) between CT in level 1 and 5 (Table 5), being their difference in temperature of 1.6°C. If predictive models were accurate, this temperature increase, or even higher, could be reached in the next years.

ANOVA results comparing canopy transparency among plots with different levels of Mean August temperature also showed significant ($p < 0.05$) differences (Fig. 3, Bottom). Bonferroni multiple comparison test revealed differences between levels 1 and 5 (Table 6), with a variation in August temperature of 2.48°C.

Table 2 Homogeneous intervals of climatic variables obtained from plots values ranging. It's shown the range in temperature, rainfall and solar radiation among plots along the plot zone

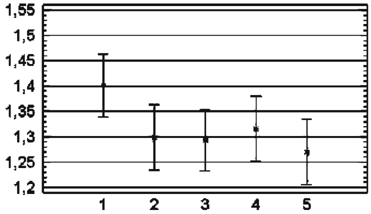
	class	Annual	December	January	February	June	July	August
Mean Temperature (ºC)	1	9 – 9.4	2.4 – 2.76	1.6 – 1.94	2.7 – 3.1	14.2 – 14.74	17.5 – 18.1	17.2 – 17.82
	2	9.4 – 9.8	2.76 – 3.12	1.94 – 2.28	3.1 – 3.5	14.74 – 15.28	18.1 – 18.7	17.82 – 18.44
	3	9.8 – 10.2	3.12 – 3.48	2.28 – 2.62	3.5 – 3.9	15.28 – 15.82	18.7 – 19.3	18.44 – 19.06
	4	10.2 – 10.6	3.48 – 3.84	2.62 – 2.96	3.9 – 4.3	15.82 – 16.36	19.3 – 19.9	19.06 – 19.68
	5	10.6 – 11	3.84 – 4.2	2.96 – 3.3	4.3 – 4.7	16.36 – 16.9	19.9 – 20.5	19.68 – 20.3
Maximum Temperature (ºC)	1	15.0 – 15.46	6.4 – 6.8	5.7 – 6.1	7.1 – 7.68	20.7 – 21.42	25.4 – 26.08	24.9 – 25.64
	2	15.46 – 15.92	6.8 – 7.2	6.1 – 6.5	7.68 – 8.26	21.42 – 22.14	26.08 – 26.76	25.64 – 26.38
	3	15.92 – 16.38	7.2 – 7.6	6.5 – 6.9	8.26 – 8.84	22.14 – 22.86	26.76 – 27.44	26.38 – 27.12
	4	16.38 – 16.84	7.6 – 8.0	6.9 – 7.3	8.84 – 9.42	22.86 – 23.58	27.44 – 28.12	27.12 – 27.86
	5	16.84 – 17.3	8.0 – 8.4	7.3 – 7.7	9.42 – 10	23.58 – 24.3	28.12 – 28.8	27.86 – 28.6
Minimum Temperature (ºC)	1	2.8 – 3.18	-1.7 – -1.36	-2.7 – -2.38	-2.1 – -1.76			
	2	3.18 – 3.56	-1.36 – -1.02	-2.38 – -2.06	-1.76 – -1.42			
	3	3.56 – 3.94	-1.02 – -0.68	-2.06 – -1.74	-1.42 – -1.08			
	4	3.94 – 4.32	-0.68 – -0.34	-1.74 – -1.42	-1.08 – -0.74			
	5	4.32 – 4.7	-0.34 – 0	-1.42 – -1.1	-0.74 – -0.4			
Rainfall (mm)	1	526.5 – 609.9	574 – 684.4	487 – 586.2	449 – 521	429 – 461.6	227 – 244.6	204 – 220
	2	609.9 – 693.5	684.4– 794.8	586.2 – 685.4	521 – 593	461.6 – 494.2	244.6 – 262.2	220 – 236
	3	693.5 – 776.9	794.8 – 905.2	685.4 – 784.6	593 – 665	494.2 – 526.8	262.2 – 279.8	236 – 252
	4	776.9 – 860.4	905.2 – 1015.6	784.6 – 883.8	665 – 737	526.8 – 559.4	279.8 – 297.4	252 – 268
	5	860.4 – 943.9	1015.6 – 1126	883.8 – 983	737 – 809	559.4 – 592	297.4 – 315	268 – 284
Solar Radiation10Kj/ m2*day*µm	1	2030 – 2046.2						
	2	2046.2 – 2062.4						
	3	2062.4 – 2078.6						
	4	2078.6 – 2094.8						
	5	2094.8 – 2111						

Table 3 Results of ANOVA between plot canopy transparency and five homogeneous levels of different climate variables

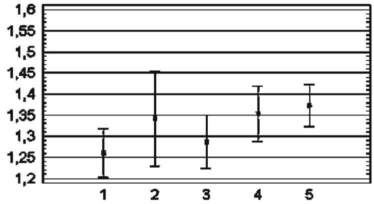
	Annual	December	January	February	June	July	August
Mean Temp.	4.200	2.231	1.535	1.873	2.106	1.282	
	0.004*	0.076	0.203	0.126	0.090	0.286	0.036*
Max. Temp.	2.318	1.492	2.029	2.275	1.345	1.466	2.281
	0.067	0.215	0.101	0.071	0.263	0.223	0.070
Min. Temp.	1.952	1.878	1.979	1.538			
	0.113	0.125	0.108	0.202			
Rainfall	2.049	2.297	1.236	1.010	1.536	**2.758**	1.519
	0.098	0.069	0.305	0.409	0.202	0.035*	0.207
Solar Rad.	2.246						
	0.091						

*Number showed F value (up) and p-value (down). Numbers in bold refer to p-values lower than 0.05. f.d. = 67 for all ANOVA

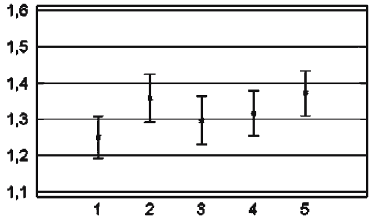
Abbreviations: Temp. = temperature; Max. = maximum; Min. = minimum; Rad. = radiation



July rainfall



Mean annual temperature



Mean August temperature

Fig. 3 Confidence intervals of ANOVA analysis among the logarithm plot canopy transparency and July rainfall (Top), mean annual temperature (Medium), and mean August temperature levels (Bottom). Each variable was categorized in five homogeneous intervals

Table 4 ANOVA between logarithm of canopy transparency and 5 July rainfall levels (Up). There were significant ($p < 0.05$) differences between CT in level 1 and 5, as it is showed by Bonferroni multiple test (Down)

July rainfall	Count	Mean	Homogeneous groups
1	13	1.27036	**a**
2	15	1.29274	**ab**
3	13	1.2988	**ab**
4	13	1.31593	**ab**
5	14	1.40098	**b**

Table 5 Bonferroni multiple test comparison demonstrated significant differences in pot canopy transparency between levels 1 and 5 of mean annual temperatures

Mean annual temperature	Count	Mean	Homogeneous groups
1	13	1.23130	**a**
2	17	1.28347	**ab**
3	18	1.34915	**b**
4	16	1.35964	**b**
5	4	1.41152	**b**

Table 6 Tukey multiple comparison test, showing significant differences in plot canopy transparency between levels 1 and 5 of mean August temperatures

Mean August temperature	Count	Mean	Homogeneous groups
1	16	1.25000	**a**
2	12	1.29750	**ab**
3	14	1.31658	**ab**
4	12	1.35859	**ab**
5	14	1.37184	**b**

The ANOVA analysis showed that there were no significant differences in CT values neither among levels of mean or minimum temperatures (annual, June, July, December, January and February) nor among different solar radiation or rainfall levels (annual, June, August, December, January and February).

To assess the influence of these parameters in plot canopy transparency, a simple regression was done for each of the significant parameters. The regression models for CT versus July rainfall, maximum temperature or August temperature were significant ($p < 0.05$), with a negative slope for the precipitation model and positive for the temperature models. The R^2_{adj} for these models was only 10.66% for July rainfall, 9.19% for Maximum temperature and 8.01% for August temperature (Fig. 4), suggesting that canopy transparency was not only explained by climatic parameters, however there was an evident influence of these parameters on this transparency.

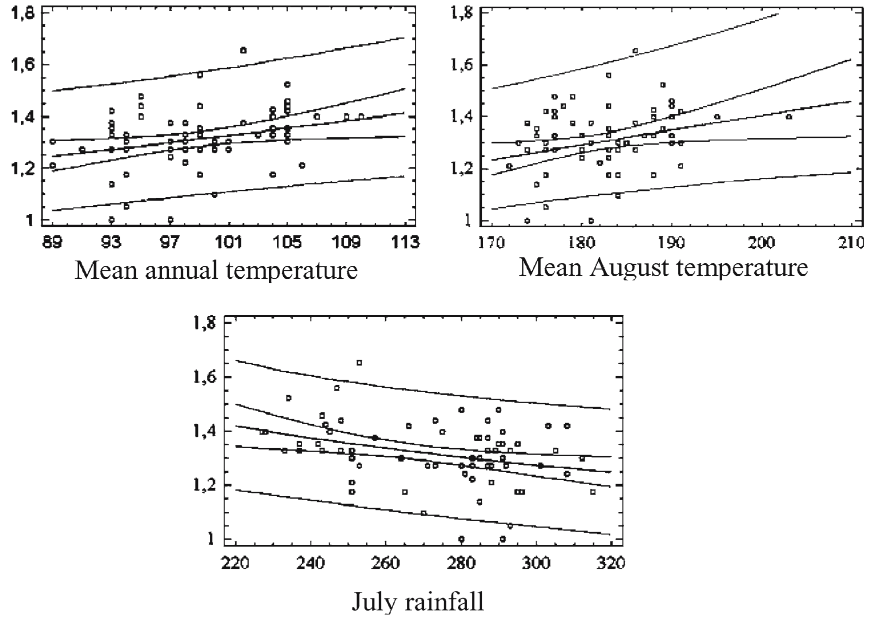


Fig. 4 Regression models for canopy transparency and climatic parameters (mean annual and mean august temperatures and July rainfall). Reciprocal-Y regression model related to maximum temperature [Plot CT = 1/(1.42482 – 0.000412949 * Maximum temperature)] (Left). Double reciprocal model related to mean August temperature [Plot CT = 1/0.155929 + 111.238/August temperature] (Right). Reciprocal-X regression model related to July rainfall [Plot CT = 0.872209 + 120.684/July rainfall] (Center)

Multiple regression analysis with backward selection showed that July Rainfall was the variable with higher effect on canopy transparency. However, although the model is significant, R^2 was quite low, with only 10.66%. Multiple regression model obtained is not improving R^2 of models with only one climatic variable.

4 Discussion

4.1 Current State

According to ANOVA results, a negative deviation in July rainfall (60 mm) led to a significant increase of CT (about 10%). Precipitation started to decrease in June, but temperatures remained at moderate levels. In July temperatures were higher but rainfall was much lower, so the combination of drought and high temperatures creates a stressful environment, causing the decrease of tree vigour. Thus, July rainfall may act as a key factor to tree condition in Castile and Leon pine forests, mainly regarding tree vigour and resistance to pest and disease. In this sense, July is not a suitable month for some management practices, as thinning and pruning in the pine forests of Castile and Leon, since the risk of insects attack will increase with the decreasing of tree vigour.

On the other hand, mean temperatures also influenced CT. As it was showed by the ANOVA analysis, an increase on this parameter (0.4°C) caused a significant increase in plot CT (6%), and an increase of 1.6°C (between levels 1 and 5) caused a CT increase of 9%, and it is just the increase of temperatures in the Iberian Peninsula over the last hundred years (Hulme & Sheard, 1999). Plots with higher mean temperatures reached higher CT values, mainly when rainfall decrease was more acute. In addition, mean August temperature also was influencing canopy, as there was an increase in CT (about 6%) when august temperature increased 2.48°C. After the drought of July, August rainfall remained very low, and its high temperature could have enhanced the effect of dry conditions, leading to a decrease of tree vigour.

Also, these variations in climatic factors may affect forest pathogens, mainly to their sporulation and colonization success, since these are influenced by changes in temperature, precipitation, soil moisture and relative humidity (Brasier, 1996; Lonsdale & Gibbs, 1996; Houston, 1998).

A pattern in temperature and precipitation was observed in relation to plot CT in the pilot zone. Thus, most of plots that were placed in warmer and drier conditions showed higher CT values, and those that were sited in colder and wetter places showed lower CT values. This agrees with results of the Programme for the Intensive Monitoring of Forest Ecosystems in Europe, that showed that between 30–50% of the variation in defoliation could be explained by the variation in stand age, soil type, precipitation, N and S deposition and foliar chemistry, for pine, oak and beech. For Scots pine, only age, precipitation and foliar Nitrogen content showed a significant relationship with defoliation, and a model with these predictor variables fitted 21% (R^2_{adj}) (De Vries et al., 2003). The main achievement of our study is the identification of mean annual and mean August temperatures as important factors affecting crown condition of pine species. It becomes clear that there are many biotic and abiotic factors affecting canopy, and meteorological factors are just a small part of them, but these factors must be included in any predictive model for forest condition.

Part of the low R^2_{adj} values from regression models could be derived from to the subjectivity of visual crown assessment. Thus, more reliable methods should be developed for crown assessment. Indirect measures such as hemispheric photography are becoming increasingly more important (Kucharik et al., 1998).

Results obtained from this study suggest that canopy transparency is influenced by temperatures and precipitations of long term preceding years, not only by recent years conditions, as it is usually thought. Historical climatic conditions may influence the present crown condition, which is a result both of recent climatic conditions and past conditions.

Although the influence of these climatic variables on canopy transparency is demonstrated, other kind of variables must be included with the aim of predicting transparency for a particular site, such as silvicultural, structural or nutritional variables. Even other factors, such as the presence of pest and diseases should be required.

4.2 Climate Change Scenarios

Climatic trends point to a likely temperature increase in south-eastern Europe of about 1–3°C (Kattenberg et al., 1996; Watson et al., 1997), and of 2°C in Spain, particularly during summer (Hulme & Sheard, 1999; Parry, 2000). Therefore, canopy transparency will follow the temperature increasing trend, causing a reduction of tree vigour, and leading to an increase of pests and diseases attack risk. If there is a 3°C increase in temperature values, canopy transparency could increase about 18%, which may represent an important reduction of canopy density. In addition, mean temperatures will probably increase due to climate change, a trend already detected in the last years in Spain (Del Río et al., 2004), so an increase of transparency is expected also. This rise in maximum temperatures may cause physiologic effects in trees, having a negative impact on primary processes as photosynthesis and causing the increase of respiration rates (Boonen et al., 2002).

In addition, rainfall will be more erratic (Dore, 2005) with a decreasing trend in Spain (Schönwiese & Rapp, 1997), so it is likely that canopy transparency rates become higher due to this decrease. In Mediterranean ecosystems, summer is a marked dry season, and at the end of the summer there is a notable reduction in the canopy. Other studies also have revealed minimum canopy cover values during summer and autumn in pine plantations (Bryant et al., 2005). Thus, it is advisable that crown assessment should be done during this period (August-September), so that canopy reduction as a result of summer drought and high temperatures will be recorded.

Forest managers must notice these trends and adapt forestry practices with the aim of minimize this defoliation effect and contribute to forest sustainability. Further surveys are required to predict defoliation values for a particular site, considering that many other kind of variables are needed for a multivariate analysis of crown condition, that would explain, which are the main factors involved in this process.

These surveys will be useful to help forest managers to minimize defoliation and to ensure the permanence of our forests in a good condition, thus helping to mitigate climate change effects over the atmosphere.

Acknowledgements Financial support of this study was provided by European Union, from the INTERREG IIIB Program, Atlantic area, exactly FORSEE Project. Thanks to Christophe Orazio for help us to obtain the climatic data and for trust us to carry out this project.

References

Brasier, C.M. (1996). *Phytophtora cinnamomi* and oak decline in southern Europe: environmental constraints including climate change. *Annales des Sciences Forestieres* 347–358.

Bonsal, B.R., Zhang, X., Vincent, L.A., Hogg, W.D. (2001). Characteristics of daily and extreme temperatures over Canada. *Journal of Climate* 14, 1959–1976.

Boonen, C., Samson, R., Janssens, K., Pien, H., Lemeur, R., Berckmans, D. (2002). Scaling the spatial distribution of photosynthesis from leaf to canopy in a plant growth chamber. *Ecological Modelling* 156, 201–212.

Brunetti, M., Maugeri, M., Nanni, T. (2000). Variations of temperature and precipitation in Italy from 1866 to 1995. *Theoretical & Applied Climatology* 65, 165–174.

Brunetti, M., Maugeri, M., Nanni, T. (2001). Changes in total precipitation, rainy days and extreme events in northeastern Italy. *International Journal of Climatology* 21, 861–871.

Bryant, M.L., Bhat, S., Jacobs, J.M. (2005). Measurements and modelling of throughfall variability for five forest communities in the southeastern US. *Journal of Hydrology* 312, 95–108.

Buffoni, L., Maugeri, M., Nanni, T. (1999). Precipitation in Italy from 1833 to 1996. *Theoretical & Applied Climatology* 63, 33–40.

Cadahia, D., Cobos, J.Mª., Soria, S., Clauser, F., Gellini, R., Grosoni, P., Ferreira, M.C. (1991). *Observación de daños en especies forestales mediterráneas*. MAPA, Secretaría General Técnica, Madrid/Comisión de las Comunidades Europeas, Bruselas. ISBN: 84-7479-880-9, 96 pp.

DeGaetano, A.T. (1996). Recent trends in maximum and minimum temperature threshold exceedences in the Northeastern United States. *Journal of Climate* 9, 1646–1660.

Del Río, S., Penas, A., Fraile, R. (2004). Analysis of recent climatic variations in Castile and Leon (Spain). *Atmospheric Research* 73, 69–85.

De Vries, W., Vel, E., Reinds, G.J., Deelstra, H., Klap, J.M., Leeters, E.E.J.M., Hendriks, C.M.A., Kerkvoorden, M., Landmann, G., Herkendell, J., Haussmann, T., Erisman, J.W. (2003). Intensive monitoring of forest ecosystems in Europe:1. Objectives, set-up and evaluation strategy. *Forest Ecology and Management* 174, 7–95.

Dore, M.H.I. (2005). Climate change and changes in global precipitation patterns: what do we know?. *Environment International* 31, 1167–1181.

Durrant, D., Boswell, R. (2002). Comparison of crown density assessments on trees within the stand and on ride edges within the forest. *Forest Ecology and Management* 157, 1–6.

EC and UN/ECE. (2000). *Forest Condition in Europe*. Executive report. UN/ECE and EC, Geneva/Brussels.

Esteban-Parra, M.J., Rodrigo, F.S., Castro-Díez, Y. (1998). Spatial and temporal patterns of precipitation in Spain for the period 1880–1992. *International Journal of Climatology* 18, 1557–1574.

Ferretti, M. (Ed.) (1994) *Especies Forestales Mediterráneas: Guía para la evaluación de las copas*. CEC-UN/ECE, Bruselas/Geneva.

Gracia, C.A., Tello, E., Sabaté, S., Bellot, J. (1999). GOTILWA: an integrated model of water dynamics and forest growth. In: Rodà, F., Retana, J., Gracia, C.A., Bellot, J. (Eds.), *Ecology of Mediterranean Evergreen Oak Forests. Ecological Studies*, vol. 137. Springer, Berlin/Heidelberg, pp. 163–179.

Heino, R., Brazdil, R., Forland, E., Tuomenvirta, H., Alexandersson, H., Beniston, M., Pfister, C., Rebetez, M., Rosenhagen, G., Rosner, S., Wibig, J. (1999). Progress in the study of climate extremes in northern and central Europe. *Climate Change* 42, 151–181.

Houston, D.R. (1998). Beech bark disease. In: Britton, K. (Ed.), *Exotic Pests of Eastern Forests*. USDA Forest service, Washington, DC, pp. 29–41.

Hulme, M., Sheard, N. (1999). *Escenarios de Cambio Climático para la Península Ibérica*. Unidad de Investigación Climática, Reino Unido, p. 6.

ICP Forests. (1992). Manual on methodologies and criteria for harmonised sampling, assessment, monitoring and analysis of the effects of air pollution on forests. *Convention on Long-Range Transboundary Air Pollution*. United Nations Environment Programme and United Nations Economic Commission for Europe, Geneva.

Innes, J.L. (1990). *Assessment of Tree Condition*. Forestry Commission Field Book 12. H.M.S.O., London, 96 pp.

Innes, J.L. (1993). *Forest Health: Its Assessment and Status*. CAB International, Wallingford.

IPCC. (2001). *Climate Change 2001: Synthesis Report*. Contribution of Working Group I and III to the Third Assessment of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge.

Karas, J. (1997). *Climate Change and the Mediterranean Region*. Report prepared for Greenpeace. Greenpeace International, Amsterdam, The Netherlands, 34 pp.

Kattenberg, A. et al. (1996). Climate models: projections of future climate. In: Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K. (Eds.) *Climate Change*, *1995: The Science of Climate Change*. Contribution of Working Group I to the Second Assessment Report of the IPCC. Cambridge University Press, Cambridge, pp. 285–357.

Klap, J.M., Oude Voshaar, J.H., de Vries, W., Erisman, J.W. (2000). Effects of environmental stress on forest crown condition in Europe. Part IV. Statistical analysis of relationships. *Water, Air, & Soil Pollution* 119, 387–420.

Kraxner, F., Nilson, S., Obersteiner, M. (2003). Negative emissions from BioEnergy use, carbon capture and sequestration (BECS)—the case of biomass production by sustainable forest management from semi-natural temperate forests. *Biomass and Bioenergy* 24, 285–296.

Kucharik, C.J., Norman, J.M., Gower, S.T. (1998). Measurements of branch and adjusting indirect leaf area index measurements. *Agricultural and Forest Meteorology* 91, 69–88.

Landscheidt, T. (2000). Solar wind near Earth: indicator of variations in global temperature. *Proceedings of the 1st Solar and Space Weather Euroconference on the Solar Cycle and Terrestrial Climate*, pp. 497–500.

Lonsdale, G., Gibbs, J.N. (1996). Effects of climate change on fungal diseases of trees. In: Frankland, J.C., Magan, N., Gadd, G.M. (Eds.), *Fungi and Environmental Change*. Cambridge University Press, Cambridge, pp. 1–19.

Mossman, V.L.F. (2002). *Anàlise especial e temporal das tendências das precipitações na Espanha Peninsular*. Doctoral thesis, 342 pp. (in Portuguese).

Ninyerola, M., Pons, X., Roure J.M. (2005). *Atlas Climático Digital de la Península Ibérica. Metodología y aplicaciones en bioclimatología y geobotánica*. Universidad Autónoma de Barcelona, Bellaterra, ISBN: 932860-8-7.

Obersteiner, M., Azar, Ch., Kauppi, P., Möllersten, K., Moreira, J., Nilsson, S., Read, P., Riahi, K., Schlamadinger, B., Yamagata, Y., Yan, J., van Ypersele, J.P. (2001). Managing climate risk. *Science* 294 (5543), 786–787.

Panel on Reconciling Temperature Observations. (2000). *Reconciling Observations of Global Temperature Change*. Climate Research Committee, Board on Atmospheric Sciences and Climate, Commission on Geosciences, Environment and Resources, National Research Council. National Academy Press, Washington, DC, 104 pp.

Parmesan, C., Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *Nature* 421, 37–42.

Parry, M.L. (Ed.) (2000). *Assessment of Potential Effects and adaptations for Climate Change in Europe: Summary and Conclusions*. Jackson Environment Institute, University East Aglia, Norwich, 24 pp.

Piervitali, E., Colacino, M., Conte, M. (1997). Signals of climatic change in the central-western Mediterranean basin. *Theoretical & Applied Climatology* 58, 211–219.

Schönwiese, C.D., & Rapp, J. (1997). *Climate Trend Atlas of Europe Based on Observations, 1891–1990*. Kluwer, Dordrecht, The Netherlands, p. 228.

Soon, W., Baliunas, S., Posmentier, E.S., Okeke, P. (2000). Variations of solar coronal hole area and terrestrial lower tropospheric air temperature from 1979 to mid-1998: astronomical forcings of change in Earth's climate? *New Astronomy* 4 (8), 563–579.

Watson, R.T., Zinyowera, M., Moss, R.H., Dokken, D.J. (1997). *The Regional Impacts of Climate Change: An Assessment of Vulnerability*. Summary for Policymakers. IPCC Working Group II Special Report, 16 pp.

Wellburn, A. (1994). *Air Pollution and Climate Change, The Biological Impact*. Longman Scientific & Technical, Essex.

Zierl, B. (2002). Relations between crown condition and ozone and its dependence on environmental factors. *Environmental Pollution* 119, 55–68.

Zierl, B. (2004). A simulation study to analyse the relations between crown condition and drought in Switzerland. *Forest Ecology and Management* 188, 25–38.

Part IV
Economic and Management Impacts

Influence of Carbon Sequestration in an Optimal Set of Coppice Rotations for Eucalyptus Plantations

L. Diaz-Balteiro[1]* and L. C. E. Rodríguez[2]

1 Introduction

The problem of determining the optimal harvest ages and the optimal number of harvests before a forest stand is re-established for coppicing tree species has been called "the coppice problem" (Medema & Lyon, 1985; Tait, 1986). The determination of optimal forest rotation ages when considering both timber production and carbon uptake is a problem that has been addressed in forest literature since the 1990s. Several authors have used some variants of the Faustmann formula to devise methods in order to incorporate the benefits of capturing CO_2.

1.1 Optimal Coppice Rotation Definition

Medema & Lyon (1985) developed an iterative procedure, based on the Faustmann solution, for the calculation of the optimal harvest ages and the number of harvests in the coppice regime. Tait (1986), working with the same example used by (Medema & Lyon, 1985), expressed the solution to the coppice problem as a recursive function and used dynamic programming to solve it. Chang (1998) outlined a generalized solution to the Faustmann's paradigm using a dynamic programming approach, following Tait (1986), which works for coppice problems, although focused mainly on non-coppicing tree species. Regarding other non-timber objectives, Smart and Burgess (2000) have blended the ideas of Chang (1998) and Tait (1986) in a recursive model in which diverse non-timber outputs (related with various flora and fauna, and benefit associated with riparian zones) are represented. Finally, Langholtz et al. (2005) studied the optimal management of a coppicing species (*Eucalyptus grandis* Hill ex Maiden) for dendroremediation purposes.

---

[1] Associate Professor at Department of Forest Economics and Management, ETS Ingenieros de Montes, Ciudad Universitaria s/n 28040 Madrid, Spain, *luis.diaz.balteiro@upm.es

[2] Associate Professor at Escola Superior de Agricultura "Luiz de Queiroz", University of São Paulo, Av.: Pádua Dias, 11 CP 9, Piracicaba – SP 13418-900, Brazil, lcer@esalq.usp.br

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

1.2 Optimal Rotation with Carbon Sequestration

The net present values of two simultaneous outcomes – timber and CO_2 uptake – were considered by Hoen and Solberg (1994) at a forest level analysis to create a production possibility frontier. The marginal cost, measured as the change in the net present value, was used for the ranking of alternative projects and the concept was applied in a case study where forest biomass growth and decay is estimated to a long-range forest management planning model. Alternative stand regimes were simulated and the forest management problem was solved by a model I type linear programming formulation. Optimal ages were determined from a constrained set of alternatives, after imposing a certain production flow and limiting the set of possible rotations generated by the forest management simulation process.

One seminal work published by van Kooten et al. (1995) studied the influence of carbon uptake on the rotation age when the prices for the carbon fixed from the atmosphere and the carbon released when the timber is harvested are equal and predetermined. The empirical results for certain values of three essential parameters – fraction of harvested timber that goes to long-term storage in structures and landfields (1/2), price of carbon that is removed from the atmosphere ($20/t), and net price of timber per unit of volume ($15/m^3) – showed increased rotation ages by approximately 20% over the level where no carbon costs or benefits were considered.

In Romero et al. (1998) a compromise programming methodology to determine optimal forest rotation ages is presented in the context of multiple uses in order to remove the divergence between private (where the net worth reaches a maximum) and social optima (where the carbon uptake reaches a maximum). The study could evaluate the influence on rotation ages when social optima and efficient allocation of resources are oriented by public financial aid, subsidies to encourage tree growth and taxes to discourage harvests.

Most of the papers that have estimated the economics of carbon dioxide sequestration by forestation activities have treated the cost of carbon sequestration as the opportunity costs of alternative land uses and the landowner's behaviour. Recent surveys covering these studies can be found in Richards & Strokes (2004) and in van Kooten et al. (2004).

The prospective benefits of carbon sequestration by forest plantations in the presence of subsidies, and the effect on the definition of the optimum rotation age of two different species (*Populus sp.* and *Pinus radiata* D. Don), were studied in Spain by Diaz-Balteiro & Romero (2003). Van Kooten et al. (1999) estimated the optimal economic rotation for four scenarios, regarding carbon account (total or annualized), discounted rate and the storage in wood products, assuming some social benefits due to carbon sequestration. Another analysis that takes into account carbon sequestration when the optimal management of slash pine plantations (*Pinus elliottii* Engelmann) involves intensive management, weed control, bedding and fertilization was provided by Stainback & Alavalapati (2005).

The economic aspects of considering carbon sequestration in Eucalyptus plantations to find the optimal number of multiple rotations have been analysed recently, although completely ignoring coppicing. Cacho et al. (2003), for example, have tested how the optimal rotation of *Eucalyptus nitens* (Deane & Maiden) Maiden plantations vary when carbon credits are included (ton-year accounting, ex-ante full crediting and ex-post full crediting). Other authors (Whittock et al., 2004a, b) have considered genetic gains in order to increase carbon sequestration in *Eucalyptus globulus* Labill plantations.

In this chapter the optimal coppice management regime considering carbon sequestration is determined for two fast growing plantation cases in Brazil and Spain using a dynamic programming approach. The analysis covers two scenarios (deterministic and non-deterministic), and three basic situations (cases) for each scenario and country. The Monte Carlo methodology has been employed to deal with the uncertainty associated with three parameters, the discount rate and the prices for pulpwood and carbon. The three basic situations are: (i) non payment for carbon sequestration, (ii) payment for carbon sequestration services including the assumption that all sequestered carbon is re-emitted at harvest time, and (iii) payment for carbon sequestration with the assumption that 50% of the carbon is re-emitted at harvest time and the rest of the re-emission occurs linearly over a time span of 5 years after harvest.

2 Materials and Methods

2.1 Study Cases

In this study, coppicing and non-coppicing management techniques are taken at constant technological levels, and plantations are managed to produce exclusively pulpwood. Clear cuts and replanting can be accomplished in one or less than 1 year. Carbon emissions during the first 2 years, when undesired sprouts are eliminated and only one or two sprouts per stump are selected, were considered negligible and it was assumed that the basic wood density does not vary over rotations. Forest carbon content due to annual biomass growth is computed as carbon captured per year.

Two different management systems for Eucalyptus plantations, one in Brazil and another in Spain, are considered to illustrate the analysis of the most important issues regarding the definition of the economical rotation that simultaneously optimizes timber production and carbon sequestration. In both countries these plantations provide the most efficient forestry systems for pulp production supplying the industry with millions of cubic meters of timber per year.

2.1.1 Brazil

In Brazil approximately 3.4 million hectares of eucalypts are intensively managed for wood production and represent 65% of the total area of planted forests (ABRAF, 2006). Among other uses, Brazilian Eucalyptus plantations are the main source of fibre for the pulp industry, which has an installed production capacity of 7.5 millions of metric tons of pulp per year, and an important renewable source of energy as charcoal for the iron industry, which has increasingly replaced fossil fuels. The states of São Paulo and Minas Gerais concentrate 55% of all eucalyptus plantations in Brazil.

In Brazil, the coppice regeneration method is widely used to manage most of its Eucalyptus plantations producing wood for the pulp, fiberboard and charcoal industry. Most of the species planted in these forests sprout profusely from stumps just after the trees are cut. The possibility of generating multiple returns from two or more rotations in the same production cycle, along with the lower cultivation costs of trees grown from the sprouts, make the eucalypts coppice system economically attractive in Brazil (Ribeiro & Graça, 1996). Most of the coppice regimes in these plantations allow more than two sprouting cycles after the first growth period. Each one of these growth cycles, called a rotation in Brazil, are usually managed for periods of 6 to 7 years. Compared to the productivity of the first rotation, volumes produced in the following rotations can be extremely variable. As mentioned by Rodriguez (1999), decreasing productivity is very common and can be caused by inadequate management of the stumps after clear cut, inappropriate choice of species/provenance for given soil and climate conditions, and harvest technology.

Table 1 presents the annual costs per hectare for the main silvicultural practices undertaken in several different eucalyptus plantations in one of the largest pulp producer states in Brazil, the state of São Paulo (1,063,744 ha). A single growth curve was used to estimate volumes (m^3/ha) for the first growth period (first rotation) in the Brazilian study case. The growth curve was obtained after adjusting a simplified version of the Schnute model (Schnute, 1981) to data collected in several different Eucalyptus plantations in the state of São Paulo. Two empirical curves were estimated for the seedling rotation of these plantations, one for higher productive sites and another for lower productive sites. Highly productive 6 years old *Eucalyptus urophylla* plantations, with an average number of 1,660 trees per hectare, yield 530 m^3/ha (site index I, with an average height of 34 m at 6 years old), while less productive plantations with the same number of trees per hectare yield 179 m^3/ha (site index II, with an average height of 24 m at 6 years old). Volume estimates for the coppicing rotations were simply based on the first rotation previous production level and reduced by a fixed rate. For the Brazilian plantations, the basic wood density was considered to vary between 0.484 and 0.793 (Scanavaca Junior & Garcia, 2004), with an average 0.655 t/m^3.

2.1.2 Galicia

In Galicia (Northwest Spain), *Eucalyptus globulus* stands constitute the largest group among other forest plantations, with a total of 383,000 ha, supplying an ever-increasing demand now at the level of 3.5 million cubic meters per year (Riesco, 2004). These plantations produce mainly pulp for paper industries: 85% of the eucalypts production is pulpwood, and a large percentage of the land occupied by

Table 1 Management costs for eucalypts plantations in the state of São Paulo, Brazil

Year	Silvicultural treatment	Cost (€/ha)
0	Site preparation and plantation costs	639
1	Weed control, fertilization, ant control	155
2	Weed control	100
3	Weed control	95
t + 1*	Fertilization and weed/ant control	120
t + 2*	Sprout selection	150
t + 3*	Weed, ant control	95
Annual	Management costs	30

t + k*, where t = clear cut age, and k = years after clear cut

Table 2 Management costs for Eucalyptus globulus plantations in Galicia

Year	Silvicultural treatment	Cost (€/ha)
0	Site preparation and plantation costs	1,297
2	Weed control	500
4	Weed control and fertilization	575
t + 2*	Sprout selection	400
t + 4*	Fertilization	75
Annual	Management costs	20

t + k*, where t = clear cut age, and k = years after clear cut

these species corresponds to non-industrial private forests, with few industrially owned plantations. As a general rule, the Eucalyptus stands in Galicia, with the exception of industrial plantations, receive less intensive silvicultural treatments. Accordingly to several sources, the most commonly accepted mean annual increment for bluegum eucalyptus in Galicia is approximately 10–15 m^3/ha/year. These mean productivities are low due to insufficient fertilization, scarce site preparation, no regard for adequate stand densities, harvests at non-optimal rotation ages, no insects or pest control, insufficient planning for adequate land use, especially in the case of small operations, and deficient wildfire management. It is also important to note that a clear cut takes place only at the end of each rotation, with no pruning, thinning or any other density lowering treatments. Galician yields used in this analysis were taken from yield tables cited by Madrigal et al. (1999) for two different site index curves, I and III. Site index I plantations, using a 1.8 × 1.8 m plantation grid, yield 554.2 m^3/ha at 15 years, and site index III plantations, using the same plantation grid, yield 276.5 m^3/ha. Management costs for Brazilian and Galician plantations are shown in Table 1 and in Table 2, respectively.

For the Spanish plantations, total carbon annually captured was computed considering a mean wood basic density of 0.5 t/m^3 (Fernández Martínez, personal communication, 2005).

In order to calculate the carbon content of each species, a 0.25 conversion factor (wood density × proportion of carbon in pulpwood) was used for the Spanish plantations. In the Brazilian plantations, this figure is 0.3275. Only the aboveground carbon (stem and aboveground biomass carbon) has been computed. To compute carbon associated to aboveground biomass, first the stem volume is estimated and then this value is multiplied by a biomass factor expansion. These biomass factors are set at 1.5 for Brazilian plantations (Penman et al., 2003) and 1.4 for Spanish plantations (Gracia et al., 2004). It was also assumed that 50% of the carbon accumulated at harvest time becomes stored after harvest as paper products, which is re-emitted lineally in the 5 years that follows clear cut (Bateman & Lovett, 2000). The other 50% were divided in two components, half is oxidized during removal, and the remaining half is slowly and linearly re-emitted in the subsequent 50 years after clear cut (van Kooten et al., 1999). Neither recycling of paper products nor changes in carbon content due to fertilization has been considered. A reference price of 10€/t of carbon captured was used in the analysis followed by a sensitivity analysis. Carbon monitoring costs and other transactional costs pointed out as important by some authors (Robertson et al., 2004) were considered negligible in this analysis. Table 3 presents a summary of all important coefficients used in the evaluations.

For the deterministic scenario, it was assumed that the wood produced in the Brazilian plantations is sold at a fixed stumpage price of 12€/m^3 (no deduction or premium paid for diameter sizes), while in the Spanish case, the pulp price considered was 38€/t for logs from trees with a minimum diameter greater than 14 cm, and 14€/t for trees with smaller diameters. Under this scenario, the discount rate imposed in the analysis was 8% in the Brazilian eucalypts, and a 5% in the Spanish plantations.

In the Spanish case, to produce the non-deterministic scenario, an exponential model was adjusted to predict wood prices based on a historical series of real prices from the year 1999 to year 2006; a triangular distribution, with average 5% and limits 3% and 7%, was used to generate discount rates; and carbon prices followed an exponential curve, with a mean of 13.30€/t, a minimum value of 6.6€/t and a standard deviation of 6.71. These data are derived from the series of values taken at the beginning of each of the last 33 months stored in the Point Carbon database.

Table 3 Summary of assumptions for the analysis

Parameter	Brazil	Galicia
Stumpage wood price, diameter < 14 cm (€/m^3)	12	14
Stumpage wood price, diameter > 14 cm (€/m^3)	12	38
Discount rate	8%	5%
Basic wood density (t/m^3)	0.655	0.500
Proportion of carbon in pulpwood	0.5	0.5
Biomass factor expansion	1.5	1.4
Carbon price (€/t)	10	10
Number of clear cuts allowed	1–4	1–4
Minimum and maximum rotation ages (years)	5–9	13–18
Possible dynamic programming states	780	1,554
Volume estimates for the coppicing rotations		
First coppicing estimate	$V_1 = 0.90^* V$	$V_1 = 1.25^* V$
Second coppicing estimate	$V_2 = 0.85^* V_1$	$V_2 = V$
Third coppicing estimate	$V_3 = 0.80^* V_2$	$V_3 = 0.75^* V$

V = seedling rotation

In the Brazilian case, a similar set was used. A triangular distribution, with average 8% and limits 6% and 18%, was used to generate discount rates; and wood prices were generated by a normal distribution adjusted for a homogenized historical deflated series of nominal prices published from Aug/2002 to Jan/2006 for the State of São Paulo by a local monthly bulletin. The same carbon price distribution model used in the Spanish case was used in the Brazilian analysis.

2.2 Methods

To determine the optimal rotation we utilized the methodology proposed by Faustmann (1849), which defines the optimum rotation as the life of the stand for which the net present value of the underlying investment achieves a maximum value, taking into account the land rent. Following Samuelson (1976), this land rent can be introduced in two different modes: supposing the existence of an infinite series of rotation cycles, or introducing the land rent explicitly in the corresponding equation. Following the first approach, and including the value of carbon sequestration (carbon stored in biomass, and carbon released after the clear cuts), the financial optimal rotation is the age at which the land expected value (LEV) reaches a maximum. LEV is calculated as follows:

$$LEV = \frac{\sum_{z=t_k}^{z=T} NPV_z - \sum_{z=t_k}^{z=T} RE_z}{1 - e^{-i \cdot T}} \tag{1}$$

with:

$z \in T$

$T = \Sigma t_k$

where t_k is the harvest age at each rotation, NPV_z is the net present value at each clear cut of trees grown from seedlings or sprout shoots, and revenues are generated by selling pulpwood and carbon credits (see Eq. 2). The number of clear cuts before a new coppice cycle begins (z) can vary between from 1 to 4. The possible rotation ages considered (t_k) are different in Brazil (5–9 years) and in Spain (13–18 years). RE_z is the sum of the discounted carbon sequestered in the biomass and released after the cutting age. Finally, T is the optimal coppice cycle (sum of the different seedling and coppice rotations ages until the removal of the old sprouts and renewal of the plantation). The following expressions detail the elements of Eq. 1:

$$NPV_z = I(t_k) \cdot e^{-i.t_k} - K - G \cdot \frac{e^{-i \cdot 1} \cdot (e^{-i \cdot t_k} - 1)}{(e^{-i \cdot 1} - 1)} - \sum_{t_j \in t_k} Y_j \cdot e^{-i \cdot t_j} - \sum_{t_j = 1}^{t_j = t_k} I_C(t_j) \cdot e^{-i \cdot t_j} \tag{2}$$

with:

$I_C(t_j) = Pc \bullet \Delta Vb_j$

$\Delta Vb_j = \rho \bullet Cc \bullet f \bullet \Delta V(t_j)$

where $I(t_k)$ corresponds to total income obtained in year t_k as the result of the wood harvested in each clear cut (independent of the tree being a seedling or a sprout shoot), K represents the plantation costs, G accounts for general annual management costs, and Y_j is the sum of the silvicultural treatments incurred after plantation or sprouting. The last term, $I_C(t_j)$, gives the discounted revenues of carbon sequestration. This revenue is the sum of the annual growth in aboveground biomass (ΔVb_j) multiplied by the carbon price (P_c). The increase in the aboveground biomass is calculated as the product of the annual growth in timber volume ($\Delta V(t_j)$) and three parameters: wood density (ρ), the carbon content (C_c), and the factor of biomass expansion (f). The discount rate is represented by i and is applied for timber and carbon benefits. Finally, modifications in some terms of Eq. 2 were introduced depending on the growth period (seedling rotation or coppicing rotation).

$$RE_z = \sum_{l=1}^{l=50} RE_l \cdot e^{-i\cdot(t_k+l)} \tag{3}$$

with:

$$RE_l = Pc \cdot \sum p \cdot BCr$$

The carbon released at harvesting is calculated (Eq. 3) for each year after the harvest (RE_l), the biomass carbon (BCr) is multiplied by the carbon price (P_c), and by a parameter p, which varies according to the quantity of carbon released, and is a function of the three possibilities: carbon stored in products, carbon re-emitted when the biomass is burned, and carbon re-emitted in the next 50 years after harvest. Finally, RE_z is the present value of annual carbon released and discounted in the future.

Dynamic programming has been used in forest management for several decades. The first applications focuses on defining the optimal scheduling of silvicultural treatments in even-aged stands (Amidon & Akin, 1968; Schreuder, 1971), including the definition of intermediate treatments and optimal forest rotations (Borges & Falcão, 1999; Brodie et al., 1978; Filius & Dul, 1992; Haight et al., 1985; Torres-Rojo & Brodie, 1990). More recently, this procedure has also been used to deal with the problem of harvesting adjacent forest stands (Hoganson & Borges, 1998; Borges & Hoganson, 1999). Cacho et al. (2003) also applied a dynamic programming algorithm to determine, in an agroforestry system, the optimal area covered with trees and the optimal forest rotation lengths used to provide carbon sequestration services.

When a sequence of interrelated decisions is presented, dynamic programming provides a very efficient procedure to solve this kind of problem, although a generalized formulation does not exist. A general structure is commonly articulated to represent a specific problem. The basic idea behind the optimization of coppice management regimes is that when a clear cut is executed a choice emerges, independently of the stand age: clear cut and replant again, or cut and let the sprouting form a new forest. Using dynamic programming terminology, a state arises at a certain age depending on the stand characteristics (age, coppice rotation, etc.). The maximum number of rotations the landowner considers opportune for each species fixes the number of stages of the problem. The decision implies moving from one state and stage to another state in the following stage. For example, in Brazilian Eucalyptus plantations, six states have been defined (clear cutting or new sprouting cycles lasting 5, 6, 7, 8 or 9 years). For each one of the states different decisions can be taken, leading to the following stage: replant again, or allow the sprouting generate a new rotation. Obviously, this model can be extended to other situations with larger numbers of states, stages and decisions.

According to Hillier & Libiermann (1991), dynamic programming problems can be solved by a backward recursive procedure that begins at the penultimate stage and looking for the best decision, one that leads to the best value of the objective function in the last stage. Next, and through an iterative procedure, the process can be repeated from the current stage once again until the initial state is reached with all chained partial solutions pointing to the total best solution. Using the notation previously introduced, for a stage z, the land expectation value (LEV_z^*) can be represented as:

$$LEV_z^* = \max \left(LEV_z^* + (LEV^*_{z+1})\bullet e^{(-i \bullet t_k)}\right) \qquad (4)$$

Where LEV_z is the land expectation value corresponding to stage z, while LEV^*_{z+1} refers to the optimal land expectation value at stage z+1, i is the discount rate, and t_k represents the length of time between stages.

For the Brazilian plantations, clear cut were considered possible for ages varying between 5 and 9 years. Coppice cycles considered a first period of growth from seedlings to adult trees ready to be cut, followed by up to three possible coppice rotations. Given these parameters, and the total number of possible alternatives to the problem, 780 possible states arise. On the other hand, for the case of eucalypts in Spain, the same four possible stages were considered (one initial growth period from seedlings and three possible sprouting rotations afterward), with each rotation varying between 13 and 18 years. These parameters defined 1,554 possible states.

The four possible stages refer to different yields, depending on the type or growth: from seedlings or from sprouting shoots. Volume estimates for the coppicing rotations in the Brazilian eucalypts were based on the seedling plantation production and reduced by a fixed rate, independent of the site quality. These rates are: 10% less for the first coppice, 15% less for the second (taken from the first coppice) and 20% less for the third sprouting rotation (taken from the second coppice). However, based on certain empirical data in Galicia (González-Río et al., 1997), we have considered the production of the first coppice rotation to be 25% higher than the production obtained in the first growth from seedlings; in the second coppice rotation, the yield equals the first clear cut volume; and in the third coppice rotation, a drop of 25% was found when compared to the first production level.

The Monte Carlo simulation helps on mitigating the effects of the natural instabilities on real the market. Other possible alternatives to solve this problem would be a stochastic dynamic programming model with a finite Markov Chain structure. As a first approach it was discarded due to its larger operational complexity.

In order to deal with risk and uncertainity, the Monte Carlo method is frequently employed. This methodology has been used in several forest management cases. Carlsson et al. (1998) have applied the Monte Carlo simulation method to analyze the spatial patterns of different habitat types and to develop models for predicting spatial patterns. It has also been used to compare Swedish non-productive stands with a random selection of stands in several different landscapes (Ask & Carlsson, 2000), and to compute the net present value in some projects (Klemperer, 2001). On those lines, Knoke et al. (2002) have used the Monte Carlo method to generate the net present distribution values of two silvicultural alternatives under uncertain future stumpage prices. Finally, McKenney et al. (2004) has also used Monte Carlo in order to simulate the uncertainty of several parameters in a spatial model that considers the afforestation with hybrid poplar species in Canada.

The Monte Carlo method is a numerical method for solving mathematical problems by simulating the values of random variables (Sobol, 1994). The method turns the problems approachable when risk exists (the probability functions for certain variables are known) or when uncertainty prevails (probabilities are unknown). In the last case, the Monte Carlo analysis is referred to as sensitivity analysis. Probability density functions are required to model the simulation process. If the probability density functions are known, the Monte Carlo simulation starts with a random sample of values generated by the probability density functions. This process continues through multiple simulations until the model converges to some acceptable results. Thus, the final result is taken as an average over the number of observations along the overall simulations. In this analysis, the @RISK software has been used to develop the Monte Carlo simulations.

Two scenarios, deterministic and non-deterministic, and three different cases have been established for both plantations. In the first case, carbon sequestration is not included; in the second case, carbon sequestration is included, following IPCC guidelines (Penman et al., 2003), with all carbon in harvested biomass oxidized at the clear cut moment; and in the third case, carbon sequestration is also considered, but a percentage of carbon remains stored in the harvested wood products. In the non-deterministic scenario, the discount rate, the pulpwood price and the carbon price are the sources of randomness.

3 Results and Discussion

Table 4 shows the optimal coppice regimes considering the deterministic scenario. Initially, the results for the Eucalyptus plantations in Spain are commented.

When carbon sequestration is not rewarded, the same coppice management regime turns out to be optimal regardless of the site index level. Moreover, the

Table 4 Optimal coppice regimes for the deterministic scenario

Eucalyptus globulus in Spain

	Site index I					Site index III				
Case	1 rot.	2 rot.	3 rot.	4 rot.	Total	1 rot.	2 rot.	3 rot.	4 rot.	Total
1	14	15	15	0	44	14	15	15	0	44
2	14	16	15	0	45	15	16	15	0	46
3	14	15	15	0	44	15	16	15	0	46

Eucalyptus urophylla in the state of São Paulo, Brazil

	Site index I					Site index II				
Case	1 rot.	2 rot.	3 rot.	4 rot.	Total	1 rot.	2 rot.	3 rot.	4 rot.	Total
1	6	6	6	0	18	7	7	7	0	21
2	7	7	0	0	14	8	8	8	0	24
3	7	7	0	0	14	8	8	7	0	23

Case 1: No payment for carbon sequestration (baseline)
Case 2: A €10 payment for carbon sequestration with 100% re-emission at harvest age
Case 3: A €10 payment for carbon sequestration with 50% re-emission at harvest age and 50% linear re-emission of the carbon stored in the product
rot. = rotation

optimal regime is homogeneous in terms of each rotation. The result of rewarding carbon sequestration can be appreciated in site index III through a minor enlargement of the optimal coppice cycle. In the three cases, a fourth rotation was not recommended by the Faustmann criterion.

In the Brazilian plantations, the optimal regime in case 1 for site index I is 18 years (Table 4), with three identical 6 year rotations. For site index II (lower productivity site index) it is 21 years, with three 7 year rotations. These plantations show a longer optimum coppice regime in the poorer sites, independent of the case considered. In the best Brazilian site, rewarding carbon sequestration (cases 2 and 3) altered substantially the optimum when compared to a non-carbon subsidized scenario. Contrary to the Spanish plantations, the length of each cycle is more responsive to site quality. Thus, a coppice cycle with two rotations is recommended for the higher productivity site in case two and case three, while for the lower productivity site a regime with three rotations is the optimum in the three cases considered.

With reference to the value of the LEV, and assuming a deterministic scenario, in the Spanish plantations the increase was less than 4% in both cases when compared to the non-subsidized alternative in site index I (Fig. 1). Conversely, LEVs increased for site index III by 5.7% and 7.2%, for case two and case three, respectively. The Brazilian plantations present a different pattern in relation to the LEV, showing larger increases when compared to the Spanish plantations in the deterministic scenario. When compared to the non subsidized alternative, LEVs increased for case 2 by 12.3% and 19.8%, respectively, for each evaluated site index. For case 3, LEVs increased 29.9% and 44.5%, respectively.

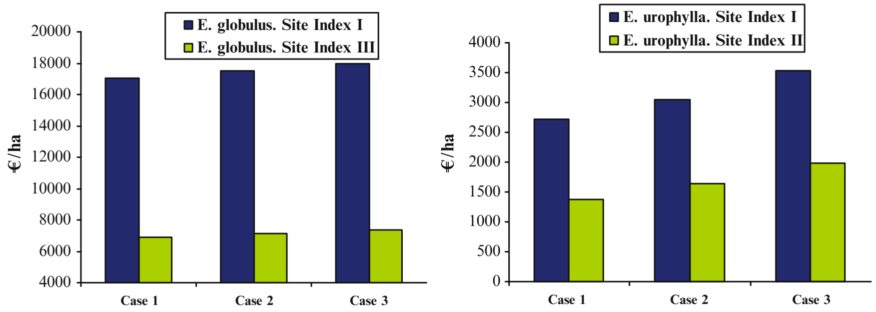


Fig. 1 LEV for both plantations. Deterministic scenario

Table 5 Optimal coppice regimes for the non-deterministic scenario

Eucalyptus globulus in Spain

	Site index I					Site index III				
Case	1 rot.	2 rot.	3 rot.	4 rot.	Total	1 rot.	2 rot.	3 rot.	4 rot.	Total
1	14	15	15	0	44	14	16	15	0	45
2	15	16	15	0	46	15	16	16	0	47
3	15	16	15	0	46	15	16	16	0	47

Eucalyptus urophylla in the state of São Paulo, Brazil

	Site index I					Site index II				
Case	1 rot.	2 rot.	3 rot.	4 rot.	Total	1 rot.	2 rot.	3 rot.	4 rot.	Total
1	6	6	6	0	18	7	7	7	0	21
2	6	6	0	0	12	7	7	7	0	21
3	6	6	0	0	12	7	7	0	0	14

Case 1: No payment for carbon sequestration (baseline)
Case 2: A €10 payment for carbon sequestration with 100% re-emission at harvest age
Case 3: A €10 payment for carbon sequestration with 50% re-emission at harvest age and 50% linear re-emission of the carbon stored in the product
rot. = rotation

Table 5 shows the results for the non deterministic scenario. The Monte Carlo simulation results were obtained after 10,000 iterations. For the Spanish eucalypts plantations, changes in the whole management cycle were very limited. Cycles got basically reduced by 1 year in most of the analyzed cases. The Brazilian analysis shows more intense differences. For case one, optimal rotations remain invariable in both deterministic and non-deterministic scenarios. When carbon sequestration is introduced in the analysis (case 2), optimal cycles get 2 or 3 years shorter, depending on to site index. In case 3, the optimal cycle reduces even more.

In contrast, LEVs from Spanish plantations result 20% to 30% less than the values obtained for the deterministic scenario (Fig. 2). These smaller values are more noticeable, *ceteris paribus*, when carbon sequestration is not considered and site index is poorer. For the Brazilian eucalypts, LEVs reflects a remarkable

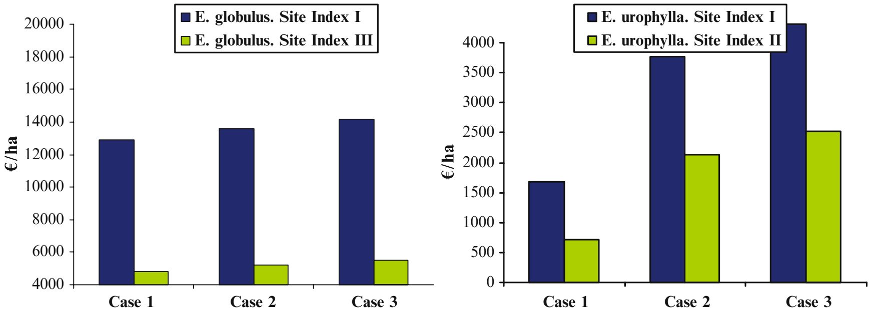


Fig. 2 LEV for both plantations. Non-deterministic scenario

decrease for both site indexes. When carbon sequestration is introduced in the analysis (case 2), there is a substantial increase in the LEV (23% to 30%). In case 3, the LEV increase is similar to case 2.

Under a deterministic scenario and in the absence of revenues and costs due to carbon sequestration (case 1), the length of the optimal coppice cycles do not vary significantly in the Spanish situation (Table 4). However, the Brazilian plantations are more sensitive to the site index or scenario. When carbon credits are included, the results suggest longer optimal coppicing rotations. It is interesting to note that in the Brazilian plantations, the optimal number of rotations was reduced by one for the best site index. Even in this case, the duration of the rotations is longer than when carbon sequestration was not rewarded.

These results coincide with the results obtained for *Eucalyptus globulus* in Portugal (Cunha-e-Sá & Rosa, 2004). Other studies for non-eucalyptus species (van Kooten et al., 1995) establish the same trend. One exception was shown by (Stainback & Alavalapati, 2005), where increases in management to promote carbon sequestration have decreased the optimal rotation length.

When randomness is introduced in the analysis (Table 5), results are quite different in the Brazilian plantations for cases 2 and 3, and a third rotation is completely disregarded in the recommendation for site index II in case 3. In the Spanish plantations the results are quite similar concerning the number of rotations per cycle and the total length of the cycle. These results are in concordance (Diaz-Balteiro & Rodriguez, 2006) if a sensitivity analysis for variations of some variables would be applied in the deterministic scenario.

If only based on Medema & Lyon (1985) and Tait (1986), decreasing rotations lengths were to be expected in cycles with multiple coppicing rotations. This was not observed in the analysis. Either for Spain and Brazil, with or without carbon subsidies, and for the whole range of productivities evaluated, shorter last rotations than the previous ones have not been the rule. The results obtained are in accordance with results reported by Smart and Burgess (2000), regarding the increase on the coppice cycle length when discount rates are higher and there carbon payments do not exist.

It is relevant to point out that the used methodology can easily incorporate different states reflecting changes on genetic material and/or silvicultural treatments to produce higher production levels (Whittock et al., 2004a,b). Biotechnological improvements and the use of cloned trees will certainly impact the definition of optimal coppice regimes. Besides, this methodology allow for the uses of different discount rates for pulp production and for carbon sequestration. Future work will also be necessary to incorporate more accurate biometric models to predict production under multi coppicing rotations, particularly in Spain.

4 Conclusions

An efficient procedure has been shown, based on dynamic programming, to determine the optimal harvest ages and the optimal number of harvests before a Eucalyptus stand is re-established when carbon sequestration is considered. The two plantations studied show different results regarding the length of the optimal coppice cycle and the land expectation value. The optimal total cycle length, though, remain relatively stable in the two scenarios considered, mainly for the Spanish plantations.

Carbon sequestration subsidies produced different results for the two cases considered. The possibility of including carbon payments impacted the revenue flow and remarkably changed optimal coppicing cycles for the Brazilian plantations (fast and short growing periods). As the land expectation value increased more than in the Spanish plantations, carbon grants are qualified to expand of the area planted with trees in Brazil and to contribute to land use changes. Finally, in the non-deterministic scenario the effects of introducing carbon payments are bigger in the LEV than in the deterministic scenario.

The inclusion of a non-deterministic scenario shows different effects (optimal cycle length and LEV) in the two plantations studied. Moreover, the results also show that the definition of the best coppicing regime can be strongly affected by the variables and parameters incorporated in the model, and that the optimal cycle can be very different from the usually recommended rotation that maximizes biomass or even the repetition of a single predetermined rotation over the whole regime.

Acknowledgements We would like to thank Dr. Manuel Fernández Martínez, of the University of Huelva (Spain) for providing important information about the basic density of *Eucalyptus globulus* in Spain. Luis Diaz-Balteiro's work was supported by the Spanish "Comisión Interministerial de Ciencia y Tecnología" (CICYT), and Luiz C. E. Rodriguez's work has been funded by the Brazilian "Conselho Nacional de Pesquisa – CNPq".

References

ABRAF (2006). Anuário Estatístico da ABRAF. Associação Brasileira de Produtores de Florestas Plantadas, Brasília, 80p

Amidon, E.L., Akin, G.S. (1968). Dynamic programming to determine optimum levels of growing stock. *For. Sci. 14*, 287–291

Ask, P., Carlsson, M. (2000). Nature conservation and timber production in areas with fragmented ownership patterns. *Forest Pol. Econom. 1*, 209–223

Bateman, I.J., Lovett, A.A. (2000). Modelling and valuing carbon sequestration in softwood and hardwood trees, timber products and forest soils. *J. Environ. Manage. 60*, 301–323

Borges, J.G., Falcão, A. (1999). Programação dinâmica e gestão de povoamentos florestais com estrutura regular e composição pura. Aplicação à Mata Nacional de Leiria. *Revista Florestal* XII, 69–82

Borges, J.G., Hoganson, H.M. (1999). Assessing the impact of management unit design and adjacency constraints on forestwide spatial conditions and timber revenues *Can. J. For. Res. 29*, 1764–1774

Brodie, J.D., Adams, D.M., Kao, C. (1978). Analysis of economic impacts on thinning and rotation of Douglas fir using dynamic programming *For. Sci. 24*, 513–522

Cacho, O.J., Hean, R.L. (2004). Dynamic optimization for evaluating externalities in agroforestry systems: an example from Australia. In: Alavalapati, J.R.R., Mercer, D.E. (Eds.) Valuing Agroforestry Systems. Kluwer, Dordrecht, The Netherlands 139–163

Cacho, O.J., Hean, R.L., Wise, R. (2003). Carbon-accounting methods and reforestation incentives. *Aust. J. Agr. Resour. Ec. 47*, 153–179

Carlsson, M., Andersson, M., Dahlin, B., Sallnäs, O. (1998). Spatial patterns of habitat protection in areas with non-industrial private forestry—hypotheses and implications. *Forest Ecol. Manag. 107*, 203–211

Chang, S.J. (1998). A generalized Faustmann model for the determination of optimal harvest age. *Can. J. For. Res. 28*, 652–659

Cunha-e-Sá, M., Rosa, R. (2004). Effects of carbon taxes and subsidies on optimal forest rotation age: an application to the Portuguese eucalyptus forest. 1st Conference of AERNA (Spanish & Portuguese Association of Environmental and Natural Resource Economics), Vigo, Spain

Diaz-Balteiro, L, Rodriguez, L.C.E. (2006). Optimal rotations on Eucalyptus plantations including carbon sequestration−A comparison of results in Brazil and Spain. *Forest Ecol. Manag. 229* (1–3), 247–258

Diaz-Balteiro, L, Romero, C. (2003). Carbon captured as a new instrument in forest management: some implications. *Scientia Forestalis 63*, 103–114

Faustmann, M. (1849). Berechung des Wertes welchen Waldboden sowie noch nicht haubare Holzbestände für die Waldwirtschaft besitzen. Allgemeine Forst und Jagd Zeitung 15. Reissued in: Faustmann, M. (1995). Calculation of the value which forest land and immature stands possess for forestry. *J. For. Econ. 1*, 7–44

Filius, A.M., Dul, M.T. (1992). Dependence of rotation and thinning regime on economic factors and silvicultural constraints: results of an application of dynamic programming. *Forest Ecol. Manag. 48*, 345–356

González-Río, F., Castellanos, A., Fernández, O., Astorga, R., Gómez, C. (1997). Manual técnico de selvicultura del eucalipto. Proxecto Columella. Escuela Politécnica Superior de Lugo, Spain

Gracia, C., Vayreda, J., Sabaté, S., Ibáñez, J. (2004). Main components of the aboveground biomass expansion factors. Cost Action EG-21 WG-1. Meeting on BEF's, Hämeelinna, Finland

Haight, R.G., Brodie, J.D., Dahms, W. (1985). A dynamic programming algorithm for optimization of lodgepole pine management. *For. Sci. 31*, 321–330

Hillier, F.S., Libiermann, G.J. (1991). Introducción a la Investigación de Operaciones. McGraw-Hill, México

Hoen, H.F., Solberg B. (1994). Potential and economic efficiency of carbon sequestration in forest biomass through silvicultural management. *For. Sci. 40*, 429–451

Hoganson, H.M., Borges, J.G. (1998). Using dynamic programming and overlapping subproblems to address adjacency in large harvest scheduling problems. *For. Sci. 44*, 526–538

Klemperer, D. (2001). Incorporating risk into financial analysis of forest management investments. In: von Gadow (Ed.) Risk Analysis in Forest Management. Kluwer, Dordrecht, The Netherlands

Knoke, T., Moog, M., Plusczyk, N. (2002). On the effect of volatile stumpage prices on the economic attractiveness of a silvicultural transformation strategy. *Forest Pol. Econom. 2*, 229–240

Langholtz, M., Carter, D.R., Rockwood, D.L., Alavalapati,J.R.R., Green, A. (2005). Effect of dendroremediation incentives on the profitability of short-rotation woody cropping of Eucalyptus grandis. *Forest Pol. Econom.* 7, 806–817

Madrigal, A., Alvárez, J. G., Rodríguez, R., Rojo, A. (1999). Tablas de producción para los montes españoles. Fundación Conde del Valle Salazar, Madrid

McKenney, D.W., Yemshanova, D., Fox, G., Ramlal, E. (2004). Cost estimates for carbon sequestration from fast growing poplar plantations in Canada. *Forest Pol. Econom.* 6, 345–358

Medema, E.L., Lyon, G.W. (1985). The determination of financial rotation ages for coppicing tree species. *For. Sci. 31*, 398–404

Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F. (2003). Good Practice Guidance for Land Use, Land-Use Change and Forestry. Intergovernmental Panel on Climate Change, Japan

Ribeiro, C.A.A.S., Graça, L.R. (1996). Manejo por talhadia: estabelecimento das idades ótimas de corte. *Revista Árvore 20*, 29–36

Richards, K.R., Strokes, C. (2004). A review of forest carbon sequestration cost studies: a dozen years of research. *Climatic Change 63*, 1–48

Riesco, G. (2004). Forest management in Eucalyptus stands: the Spanish case. International IUFRO 4.04.06 Meeting: The Economics and Management of High Productivity Plantations, Lugo, Spain

Robertson, K., Loza-Balbuena, I., Ford-Robertson, J. (2004). Monitoring and economic factors affecting the economic viability of afforestation for carbon sequestration projects. *Environ. Sci. Pol. 7*, 465–475

Rodriguez, L.C.E. (1999). Defining the optimum sequence of rotations for coppice regimes of eucalyptus. In: Chang, S.J. (Ed.) Proceedings of the International Symposium 150 Years to the Faustmann Formula: Its Consequences for Forestry and Economics in the Past, Present and Future, Darmstadt

Romero, C., Ríos, V., Diaz Balteiro, L. (1998). Optimal forest rotation age when carbon captured is considered: theory and applications. *J. Oper. Res. Soc. 49*, 121–131

Samuelson, P.A. (1976). Economics of Forestry in an evolving society. *Econ. Inq. 14*, 466–492

Scanavaca Junior, L., Garcia, J.N. (2004). Determinação das propiedades físicas e mecânicas da madeira de Eucalyptus urophylla. *Scientia Forestalis 65*, 120–129

Schnute, J. (1981). A versatile growth model with statistically stable parameters. *Can. J. Fish. Aquat. Sci. 38*, 1128–1140

Schreuder, G.F. (1971). The simultaneous determination of optimal thinning schedule and rotation for an even-aged forest. *For. Sci. 17*, 333–339

Smart, J.C.R., Burgess, J.C. (2000). An environmental economic analysis of willow SRC production. *J. For. Econ. 6*, 193–225

Sobol, I.M. (1994). A primer in Monte Carlo Method. CRC, Boca Raton, FL

Stainback, G.A., Alavalapati, J.R.R. (2005). Effects of carbon markets on the optimal management of Slash Pine (Pinus elliottii) plantations. *Southern J. Appl. For. 29*, 27–32

Tait, D.E. (1986). A dynamic programming solution of financial rotation ages for coppicing tree species. *Can. J. For. Res. 16*, 799–801

Torres-Rojo, J.M., Brodie, J.D. (1990). Demonstration of benefits from an optimization approach to the economic analysis of natural pine stands in Central Mexico. *Forest Ecol. Manag. 36*, 267–278

van Kooten, G.C., Binkley, C.S., Delcourt, G. (1995). Effect of carbon taxes and subsidies on optimal forest rotation age and supply of carbon services. *Am. J. Agr. Econ. 77*, 365–374

van Kooten, G.C., Krcmar-Nozic, E., Stennes, B., van Gorkom, R. (1999). Economics of fossil fuel substitution and wood product sinks when trees are planted to sequester carbon on agricultural lands in western Canada. *Can. J. For. Res. 29*, 1669–1678

van Kooten, G.C., Eagle, A.J., Manley, J., Smolak, T. (2004). How costly are carbon offsets? A meta-analysis of carbon forest sinks. *Environ. Sci. Pol. 7*, 239–251

Whittock, S.P., Greaves, B.L., Apiolaza, L.A. (2004a). A cash flow model to compare coppice and genetically improved seedling options for Eucalyptus globulus plantations. *Forest Ecol. Manag. 191*, 267–274

Whittock, S.P., Apiolaza, L.A., Dutkowski, G.W., Greaves, B.L., Potts, B.M. (2004b). Carbon revenues and economic breeding objectives in Eucalyptus globulus pulpwood plantations. In: Borralho, N.M.G., Periera, J.S., Marques, C., Coutinho, J., Madeira, M., Tomé, M. (Eds.) Eucalyptus in a Changing World. IUFRO Symposium, Aveiro

Use of Forests and Wood Products to Mitigate Climate Change

L. Valsta[1]*, B. Lippke[2], J. Perez-Garcia[2], K. Pingoud[3], J. Pohjola[1],
and B. Solberg[4]

1 Introduction

The increased concentrations of greenhouse gases in the atmosphere are one of the most severe current environmental problems. The annual atmospheric increase of carbon is estimated to be 3.2 Pg (IPCC, 2001, p. 190). In comparison, the annual harvest of roundwood is about 3.5 billion cubic meters (FAO, 2006) and contains approximately 0.8 Pg carbon in roundwood (assuming 0.23 Mg C/m^3) and is, hence, significant also for the global carbon balance. The estimated amount of carbon in forested areas is approximately 650–1,200 Pg (House et al., 2003; Grace, 2004; FAO, 2006), most of which is located in forest soils. Recent aboveground biomass estimates are between 257 Pg (Kauppi, 2003) and 359 Pg (IPCC, 2001). Given the large amounts, even a small proportional change is influential.

Industrial use of wood fulfills a share of the material needs of the human population. Environmental policies can influence the consumption of alternative materials. Given the large mount of global wood utilization, it is relevant to ask what are the impacts of wood use on the global carbon cycle. Should more wood be consumed to replace materials that require more fossil energy or should less wood be consumed to increase the forest carbon sink?

Wood carbon has several, partly competing, functions in climate change mitigation: (1) wood carbon can be stored into forest ecosystems by different silvicultural strategies; (2) wood carbon can be stored as products in use or in landfills; (3) wood products can be used for materials that substitute other materials with higher fossil emissions; (4) wood is used for bioenergy in different stages of the life cycle.

---

[1] Department of Forest Economics, University of Helsinki, Finland,* lauri.valsta@helsinki.fi

[2] College of Forest Resources, University of Washington, Washington, DC

[3] VTT Technical Research Centre of Finland, Finland

[4] Department of Ecology and Natural Resource Management, Norwegian University of Life Sciences, Norway

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

When wood is used for materials and energy, other materials and energy are substituted. Use of these substitutes in place of wood would in most cases cause larger fossil emissions. Valsta et al. (2005) have preliminary estimated that, given the current materials use, the global potentially avoided emissions due to wood use are around 0.4 Pg carbon. At the same time, the utilization of forest resources and land-use change cause a release of carbon of 1.1 Pg, mostly due to deforestation in the tropical areas (FAO, 2006). The regional patterns of these changes are shown in Fig. 1. The boreal and temperate forest regions exhibit an increase of biomass while a strong decline takes place in the tropical areas.

The global average annual forest biomass decline rate has slightly increased from 0.37% to 0.40% for 1990–2000 and 2000–2005, respectively. The net woody biomass increase in the temperate and boreal forest regions of the world was estimated to be 0.88 Pg/year in 2000 (FAO, 2001) and in 2005 down to 0.30 Pg/year. An even lower estimate, 0.21 Pg/year into living biomass in the Northern Hemisphere, is given by Goodale et al. (2002). A recent study by Kauppi et al. (2006) rather suggests an increasing trend in biomass accumulation into boreal and temperate forests.

According to FAO's Global Forest Resources Assessment 2005 (FAO, 2006), the global wood removals (harvests) have remained at an approximately constant level from 1990 to 2005 at 3 billion cubic meters. Of the removals, 60% were industrial

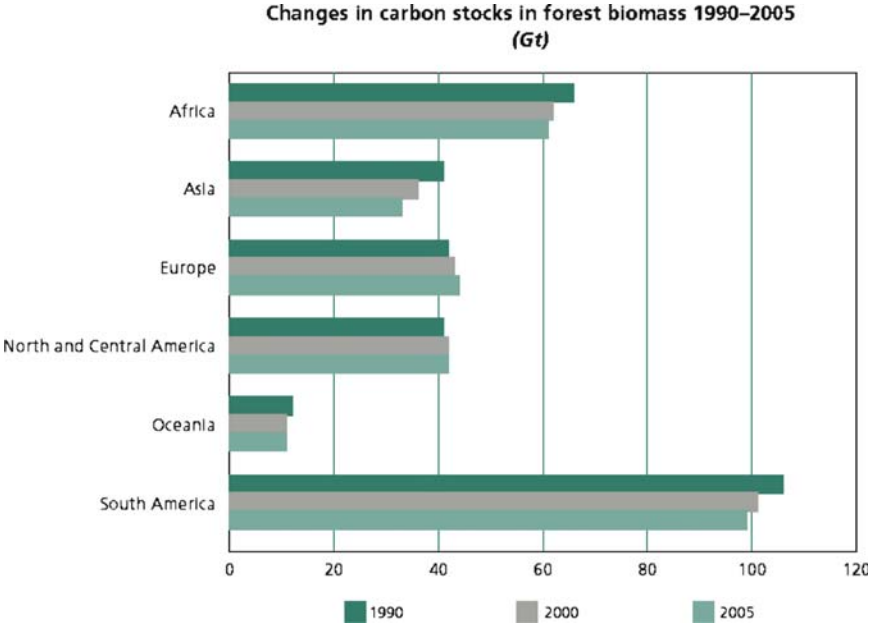


Fig. 1 Changes in regional forest biomass (FAO, 2006)

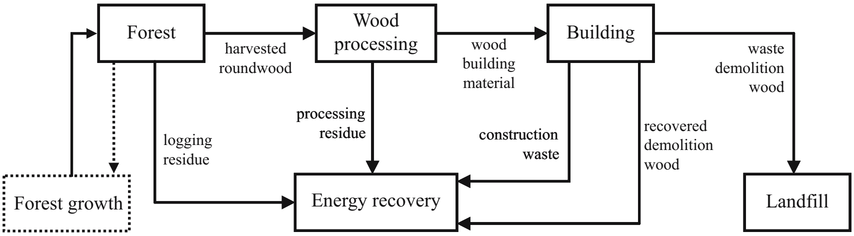


Fig. 2 Carbon flows in an integrated analysis covering forest dynamics and the wood product life cycle (Modified from Gustavsson et al., 2006)

roundwood and 40% fuelwood. However, FAOSTAT statistics (2006) report a 500 million cubic meters higher fuelwood use and the resulting total roundwood removals became 3,490 million cubic meters for 2005 with industrial roundwood and fuelwood shares of 48% and 52%, respectively.

To evaluate the flow of carbon through wood use, life-cycle analyses are required. Most life-cycle analysis studies include the wood flow from harvest to demolition. As such, they do not address the dynamic nature of forest carbon pools: the growth of forest (the flow of carbon from atmosphere into forest) dynamically depends on the size of the forest carbon pool (Fig. 2). Additionally, this relationship is nonlinear so that an increase of forest carbon pool first increases the growth but as crowding increases growth decreases. This review addresses the need for integrated analysis where forest dynamics and wood use impacts are jointly analyzed.

Most of the world's managed forests are under so called even-aged management or rotation forestry. We assume here that such forests are managed with sustained yield and regeneration of the cut areas. In these forests, an increase in carbon pools can result from either an increase in rotation length or an increase of growing density. The former is achieved just by postponing the final harvest. The latter requires either reduced thinnings or more efficient regeneration and young stand management. Integrated studies can be made where the impacts of wood use on both the forest dynamics and material life cycle are addressed. In the following, we analyze three such studies and synthesize knowledge based on them.

2 Case Studies

The case studies that we review address the impacts of forests and wood products in an integrated way. The studies have adopted different methods to analyze the question and they also have different temporal and spatial characteristics. Two of them refer to northern Europe and one to North America.

2.1 CORRIM Study (Perez-Garcia et al., 2005a)

The CORRIM studies build on a large body of life-cycle analyses of different wood products (Perez-Garcia et al., 2005b), analyzed in the context of residential construction. They build a carbon and emissions accounting model where three carbon pools were identified: carbon in forest, carbon in forest products, and carbon associated with energy displacement and avoided emissions. Starting from stand establishment, these carbon pools area tracked over time as the forest grows, wood gets harvested and processed, and products are used to build residential houses. Pulp and paper manufacturing was not considered. To evaluate the avoided emissions due to wood use, functionally equal residential houses were compared with wood vs. concrete or steel frames in structures. The wooden houses contained 1.97 times more wood than the alternative steel-framed house (Lippke et al., 2004).

To be able to compare alternative buildings, the CORRIM studies covered a detailed analysis of the production and use of wood building materials (lumber, plywood, OSB, glulam, laminated veneer lumber and I-joists). The SimaPro software was used to construct life-cycle inventories for each product. These product data were incorporated into the Athena Environmental Impact Estimator model which also contains data about alternative construction materials.

The emissions accounted for included those from silvicultural operations, harvesting stands, and manufacturing wood products. The additional biofuel substituted for natural gas.

The environmental performance of alternative materials was compared using several indices: embodied energy, global warming potential, air emission index, water emission index, and amount of solid vaste. The global warming potentials of steel vs. wood and concrete vs. wood buildings were 26% and 31% larger, respectively (Lippke et al., 2004).

To analyse the carbon dynamics over time, a combined forest, products, and substitution carbon model was build (Manriquez, 2002). Forest development was projected using the LMS system (Oliver, 1992). It accounted for tree canopies, stems, roots, litter and snags. Wood products were divided into short-term and long-term products. Different carbon pools had individual decay rates that were applied in the simulation. Four management scenarios were formed: managed Douglas fir forest with 45, 80 and 120 years rotations, and a no-harvest scenario for 165 years (the 80-year regime is shown as an example in Fig. 3). Simulations were carried out for a total time of 165 years.

Comparisons between rotation ages showed two main characteristics. For the combined carbon pools of forest and products, longer rotations increased the carbon pools, at least to the age of 165 years. Contrary, when energy and material substitution were added, the shorter the rotation, the greater the cumulative mitigation impact of the whole (Fig. 4).

With the assumption that a given house will be built with either wood frame or alternative frame, the global warming potentials (GWP) can be compared. The global warming potentials were 37,047 and 46,826 kg for wood and steel frame houses,

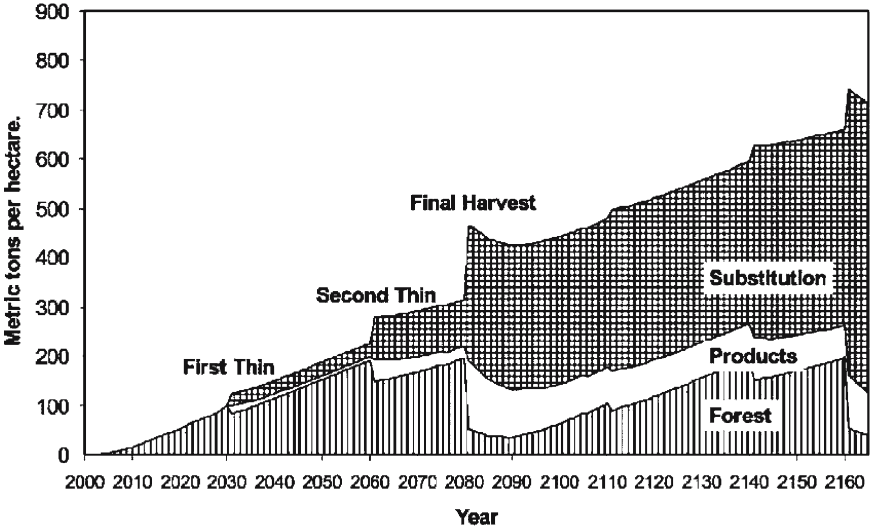


Fig. 3 Carbon in the forest and product pools with concrete substitution for the 80-year rotation (from Lippke et al., 2004)

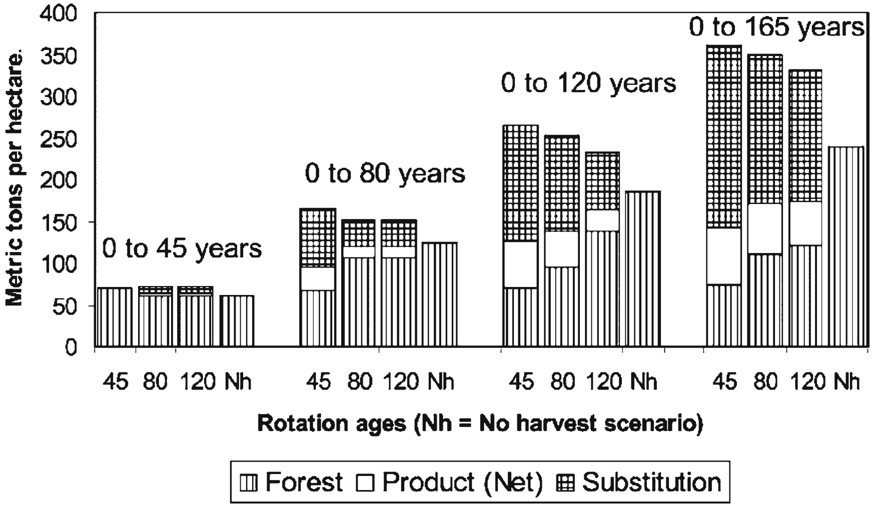


Fig. 4 Average carbon per year in forest, product, and concrete substitution pools for different rotations and simulation times

respectively (ratio 1.26) (Lippke et al., 2004). These amounts can be made relative to the carbon in wood employed in houses (6,496 and 3,298 kg, respectively). The incremental wood carbon used in a wooden house reduced the global warming potential at the rate of – 0.83 metric tons of GWP in C equivalents units for each additional metric ton of wood C in wood.

2.2 Pingoud et al., 2006

The study (Pingoud et al., 2006) examines the relationships between carbon pools in the forest and wood products use. The study compared present silvicultural guidelines in Finland to modifications that would change the climate change mitigation impact of forests. Silvicultural changes included increases of rotation length and of growing density. These alternatives have two kinds of impacts: (1) they affect wood yield and consequently the amounts of wood products; (2) they change forest carbon sequestration and the steady state carbon pool.

The study is based on a steady state analysis of fully regulated forests (also known as normal forests) with alternative silvicultural practices. The annual wood yield is constant but different in each alternative. The analysis employs a baseline alternative where a given silviculture yields corresponding amounts of wood products. On the consumption side these products are used to fulfill a functional need, in this case housing, pulp and paper manufacture and bioenergy. To compare the alternatives to the baseline, it was assumed that the same material function had to be fulfilled. For sawtimber it was assumed that the number of houses was fixed in all the alternatives but the share of wooden houses was dependent on the sawtimber yield. With a modified silviculture, the sawtimber yield is, in our simulations, increased and, consequently, more houses with wooden frames are being built. For the same simulations pulpwood yields often decreased but not always. Note that also concrete framed houses utilize significant amounts of wood products. The change in building materials led to a change in net fossil carbon emissions according to the energy usage of different building materials and to the om pulp and paper manufacture are based on the study by Pingoud and Lehtilä (2002) substitution of fossil fuels by bioenergy originated in energy wood from forest, residues from wood processing, and construction and demolition waste from housing. The data for emissions and material use are from Gustavsson et al. (2006). The data on fossil carbon emissions from pulp and paper manufacture are based on the study by Pingoud and Lehtilä (2002).

Silviculture scenarios were based on the silvicultural guidelines issued in Finland (Hyvän metsänhoidon suositukset, 2001), as realized in the Motti simulation software (Hynynen et al., 2005). In addition to the baseline, modified silvicultural regimes were:

- Increase of rotation by 20 or 40 years
- Increase of rotation by 20 or 40 years and increase of basal area by 4 m^2/ha
- Precommercial thinning of energy wood

Basal area increase was defined as an equal change in basal area before and after thinning, causing the thinning to be postponed. Comparing the alternatives we observed that at first, increasing forest biomass lead to increased growth and yield of wood products. When the biomass was further increased, growth began to decline and the wood product yield, as well.

Fully regulated forests were constructed that employed the given regimes, one each. The resulting yields of sawtimber were directed to produce sawn wood and wood-based panels to be used in house construction. For pulpwood two consumption sub-scenarios were considered both meeting the condition of the same material function: (1) pulp and paper production was constant in each alternative; in case of excess pulpwood yield it was utilized as bionergy to substitute coal, (2) pulp and paper production varied with pulpwood yield, but in case of paper deficit an emission-free substitute (e.g. electronic media) was assumed be applied on the consumption side to fulfill the same material function. From Scots pine chemical paper was manufactured, and from Norway spruce mechanical paper, both having different emission profiles. Analyses were carried out for Scots pine and Norway spruce forests, separately. Only the results for Norway spruce are presented here.

When wood materials are directed to different uses, they affect the fossil emissions, relative to the baseline. The fuel replaced by bioenergy was assumed to be coal, and oil was assumed to be used in production of building materials. The avoided emissions due to an increase in wood use can be compared to the carbon in the harvested biomass. In the article by Pingoud et al. (2006) a marginal fossil carbon substitution factor was introduced as relating the fossil C emission reductions (with respect to a baseline) to the additional wood biomass use (with respect to the wood yield in baseline). This gave relative substitution coefficients which indicate how much fossil emissions are changed for each ton of additional wood harvested to different uses (for Norway spruce):

Sawtimber (Swedish multi-story apartment house)	–2.05
Sawtimber (Finnish multi-story apartment house)	–1.31
Pulpwood (pulp & paper production constant, excess wood for bioenergy)	–0.89
Pulpwood (function constant, paper deficit replaced by electr. media)	0.48
Energywood (for bionergy to replace coal)	–0.89

Each silvilcultural regime produced a given amount of sawtimber, pulpwood, and energy wood. When their emissions are totalled, we can compute the average emissions per year and hectare for each regime. For Norway spruce, when applying the Swedish building data, increases in rotation and basal area lead to decreases in emissions (Fig. 5). The change was somewhat larger due to basal area change (defined as basal area before and after thinning), compared to rotation change. The positive substitution factor of pulpwood is related to sub-scenario (2) above. Emissions would be increased with increased pulpwood use in this latter case – or reduced by decreased consumption – because the substitute would cause less emissions than pulp and paper production. This is seen in the effects due to basal area change which increased the growth rate of stands at thinning ages.

The silvicultural regimes also differed in terms of carbon pools in the forest and products. Although there is no evident way of comparing the benefit from a change in carbon pools with the change in annual emission, we portray them side by side by computing the present value in physical terms of the tons of avoided emissions (cf. Hoen & Solberg, 1994). Figure 6 shows this comparison for the case where excess pulpwood is directed into pulping. It can be seen that the present value of emission reductions is generally slightly smaller than the change in carbon pools for Norway spruce.

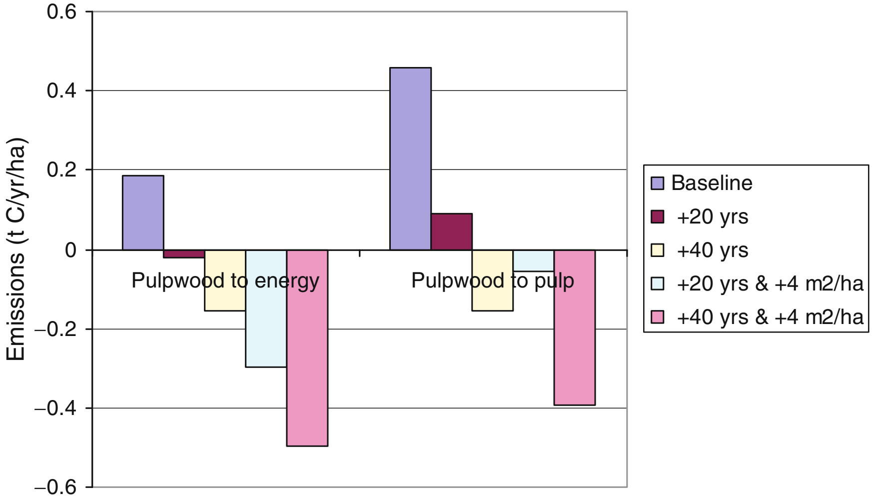


Fig. 5 Total emissions from wood utilization based on silvicultural regimes for Norway spruce, Swedish building

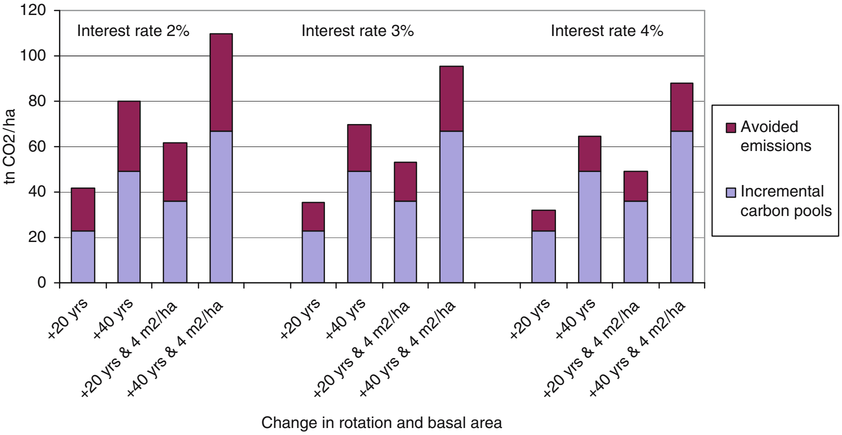


Fig. 6 Changes in the present value of discounted emissions and steady state carbon pools due to changes in rotation and stand basal area before and after thinning, Norway spruce

Compared to present silvicultural guidelines in Finland, an increase in rotation and growing density increased carbon pools in forests and wood products. They also increased the avoided emissions for Norway spruce because more wood was available for sawtimber to substitute for concrete framed houses. This result is naturally conditioned to the fixed amount of house construction. It should be noted additionally, that further increases of rotation or growing density may not show further increases in emissions reductions because forest growth might not increase any more.

2.3 GAYA-J/C Model (Petersen et al., 2004, 2005)

While the two previous studies were based on stand-level and conceptual fully-regulated forest level analyses, the GAYA/JC model operated on forest or regional level. The analyses reviewed here pertain a large forest area in Southern Norway. The model connects forest planning with climate change mitigation impacts based on forest and forest product use.

The basis of the model is a forest area planning model that utilizes forest inventory data, whole-stand growth models and forest management optimization using linear programming. Forest management activities in the model are

- No treatment
- Release thinning
- Thinning
- Fertilization
- Clear cut
- Seed tree cut
- Planting

Each of these activities have many alternatives defined, and the model solutions contain optimal harvests of pulpwood and sawtimber over time in the forest area analyzed. The carbon accounting module is comprehensive and includes trees, dead wood, litter, harvest residues, soil, wood products, and energy and material substitution. Carbon related benefits are computed as the present value of sinks (forests and wood products) and emissions reductions. Wood use is divided into energy, sawtimber, and pulp and paper.

Both economic variables and climate change mitigation impacts can be specified as objectives for the analyses. Additionally, different variables can be treated as constraints, which enables trade-off analyses between the goal variable and constraints, such as NPV of timber harvests vs. mitigation impacts. In addition to aggregate variables, the model solution identifies the optimal management regimes for forest stands (or for classes of stands) over time for the specified objective function and constraints. The model also operates with various discount rates (from 0 p.a. and upwards) reflecting the weight one puts on when in time benefits and costs occur.

The case study about the Hedmark County has the following characteristics:

- Productive forest land 1.3 million hectares
- Data from 2,207 sample plots
- Forty-seven percent Scots pine, 41% Norway spruce, 8% broadleaves in forest inventory
- Twenty-eight percent of area is in the oldest age class
- The annual actual harvest has been 2.3 million cubic meters
- The model runs span 12 planning periods, 10 years each

The starting points of the analyses were two optimization problems and their solutions: maximization of net present value of timber harvests, and maximization of

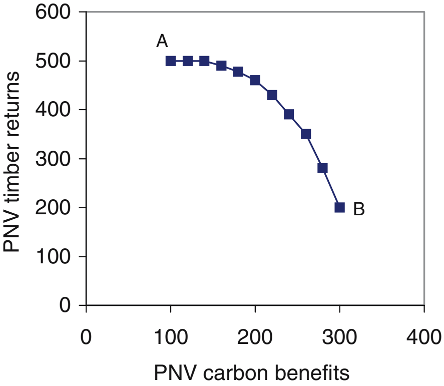


Fig. 7 Trade-off between timber returns and carbon benefits

the present value of carbon benefits (sequestration and substitution). These two solutions provided the extremes for a trade-off analysis. Intermediate optimum solutions were generated by maximizing the present value of timber returns subject to 10%…90% range of carbon benefits (their present value) (Fig. 7). When the weight of carbon benefits was increased (going from point A to point B), the frequency of harvest activities decreased leading to fewer thinnings, fewer release thinnings, and longer rotations. At the same time, more of the regenerated area was planted as opposed to natural regeneration. Average standing volume of the forest area increased very much, from 100 to 350 m^3/ha.

The model permitted also an analysis of the impacts of including energy and material substitution effects in carbon benefits. That resulted commonly a 20% to 70% increase of present net value of carbon benefits (and in some cases up to 1,900%). Forest management activities were somewhat intensified, for example planting area was increased. A larger share of wood was harvested as sawtimber.

3 Discussion

The three studies reviewed share, at large, the same components to analyze the carbon pools and substitution effects. They differ markedly in the analysis setting in terms of method and baseline. Also the biological and resource utilization patterns differ. Common to all of them is that a baseline forestry is compared to alternatives that differ in their climate mitigation impacts.

The first two studies are based on management alternatives for individual stands. The increase in rotation length brings about opposing effects: In Pingoud et al. (2006), an increase in rotation length (relative to the baseline) leads to an increase in sawtimber yields but a decrease in pulpwood yield in most scenarios and, hence, an increase in substitution effect being more pronounced for sawtimber than for energywood and pulpwood – or even negative for pulpwood, if paper could be replaced by a less-emission intensive substitute. In Perez-Garcia et al. (2005b), the outcome is different because the extensions in rotation are considerably larger, leading to significant decreases in fossil fuel displacement as bioenergy and to no change or a decrease in the amounts of structural wood products, and thereby decreases in the total mitigation effect. In the third study, rotations are generally lengthened due to increased weight on mitigation benefits (prolonged GHG emission at harvesting time and getting more timber for sawnwood) in the objective function.

Intensifying silviculture (investing more in management) relative to the baseline was favorable for mitigation in studies 1 and 3. Study 2 did not have an option to invest in more intense silviculture, but the option of postponing thinnings and increasing growing densities is quite analogous (because of additional capital investment in growing stock) and increased mitigation effect. The common outcome of the studies was that increased silvicultural input was beneficial to mitigation, given the baselines used. In Pingoud et al. (2006) maximization of the sawtimber yield appears to give the highest substitution impacts. However, the results could have been somewhat different, if pulpwood were used to produce wood-based panels for construction purposes. It is presumable that the maximum substitution impacts would then be obtained with a silviculture close to one where the total wood yield is at the maximum.

For the Nordic conditions and for both Scots pine and Norway spruce, Liski et al. (2001) report results from a 30-year shortening and lengthening of rotation from a 90-year baseline. For Norway spruce, the vegetation and product carbon pools increased with increasing rotation but it appeared that they would start to decrease some time after 120 years. On the contrary, soil carbon pools decreased significantly with increasing rotation because of strongly reduced litter flow to the soil. Due to uncertainties in soil carbon modeling, the authors overall recommended the longer rotations for climate change mitigation. As plot averages, the rotations in study 2 were 57, 77, and 97 years. For these, the results showed increasing avoided emissions with increasing rotation and were in agreement with findings by Liski et al. (2001).

The realized substitution effects also depend on the patterns of material utilization, namely which products are produced from wood. In the CORRIM study, small diameter wood and residues from sawmilling were used as short term products but not explicitly as pulp raw material. The share of wood products for building was considerably high, compared to the Nordic studies.

The increase of wood usage in wood framed houses, as compared to concrete or steel framed houses, affects the potential to increase wood use. In the CORRIM model houses, it was 20% and 97% while in the Nordic houses 40% and 350%. However, the avoided emissions relative to additional wood use do not necessarily depend on the amount of wood use increase. The Swedish house had a smaller increase of wood use but higher fossil emission reductions per ton of wood used, compared to the Finnish house.

Given the large impact of wood product use, taking into account the wood products in international climate conventions is an important question. As Niles and Schwarze (2001) note, material substitution and energy substitution are more viable long-term climate change strategies than is sequestration. Our results strongly support this conclusion.

The studies reviewed in this presentation represent boreal and moist temperate conditions. The impacts of changing the forest rotation depend on the chosen baseline. The relation between rotations that maximize (i) mean annual increment, (ii) present net value of timber returns, and (iii) climate change mitigation varies among biological and economic conditions. For example, the flow of carbon into soil is relatively inadequately assessed by current models, compared to above ground flows. Although the CORRIM study and the Petersen et al. (2004, 2005) study included models for soil carbon, the true effects of silvicultural alternatives in soil dynamics may modify the results obtained in these studies.

The integrated studies reviewed in this paper covered climate change impacts of forestry as well as energy and material substitution of wood products in boreal/temperate conditions. The impacts of wood products use significantly change the contribution of forestry to climate change mitigation: managed forests become more beneficial compared to unmanaged. Especially, production of long-lived wood products is an efficient way of mitigating climate change. This should be kept in mind when designing forest and environmental policies that direct the use of the renewable forest materials.

References

FAO (2001). Global forest resources assessment 2000. Main report. FAO Forestry Paper 140. Food and Agriculture Organization of the United Nations, Rome, 479 p.

FAO (2006). Global forest resources assessment 2005. Main report. FAO Forestry Paper 147. Food and Agriculture Organization of the United Nations, Rome, 320 p.

FAOSTAT Statistics (2006). [Referenced 15.11.2006] http://faostat.fao.org/site/513/DesktopDefault.aspx?PageID=513.

Global Forest Resources Assessment (2005). FAO Forestry Paper 147. Food and Agriculture Organization of the United Nations, Rome, 320 p.

Goodale, C.L., Apps, M.J., Birdsey, R.A., Field, C.B., Heath, L.S., Houghton, R.A., Jenkins, J.C., Kohlmaier, G., Kurz, W., Liu, S., Nabuurs, G.-J., Nilsson, S. and Shvidenko, A.Z. (2002). Forest carbon sinks in the Northern Hemisphere. *Ecological Applications 12*(3):891–899.

Grace, J. (2004). Understanding and managing the global carbon cycle. *Journal of Ecology 92*:189–202.

Gustavsson, L., Pingoud, K. and Sathre, R. (2006). Carbon dioxide balance of wood substitution: Comparing concrete and wood-framed buildings. *Mitigation and Adaptation Strategies for Global Change 11*:667–691.

Hoen, H.F. and Solberg, B. (1994). Potential and economic efficiency of carbon sequestration in forest biomass through silvicultural management. *Forest Science 40*(3):429–451.

House, J.I., Prentice, I.C., Ramankutty, N., Houghton, R.A. and Heimann, M. (2003). Reconciling apparent inconsistencies in estimates of terrestrial CO2 sources and sinks. *Tellus* B *55*(2), 345–363.

Hynynen, J., Ahtikoski, A., Siitonen, J., Sievänen, R. and Liski, J. (2005). Applying the MOTTI simulator to analyse the effect of alternative management schedules on timber and non-timber production. *Forest Ecology and Management 207*:5–18.

Hyvän metsänhoidon suositukset (2001). Metsätalouden kehittämiskeskus Tapio. Helsinki, 95 p.

IPCC (2001). In: Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J., Dai, X., Maskell, K. and Johnson, C.A. (eds.). Climate change 2001: The scientific basis. Contribution of the Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, 881 p.

Kauppi, P.E. (2003). New, low estimate for carbon stock in global forest vegetation based on inventory data. *Silva Fennica 37*(4):451–457.

Kauppi, P.E., Ausubel, J.H., Fang, J., Mather, A.S., Sedjo, R.A. and Waggener, P.E. (2006). Returning forests analyzed with the forest identity. *Proceedings of the National Academy of Sciences of the United States of America 103*(46):17574–17579.

Lippke, B., Wilson, J., Perez-Garcia, J., Bowyer, J. and Meil, J. (2004). CORRIM: Life-cycle environmental performance of renewable building materials. *Forest Products Journal 54*(6):8–19.

Liski, J., Pussinen, A., Pingoud, K., Mäkipää, R. and Karjalainen, T. (2001). Which rotation length is favourable to carbon sequestration? *Canadian Journal of Forest Research 31*(11):2004–2013.

Manriquez, A.C. (2002). Carbon sequestration in the Pacific Northwest: A model. Master of Science thesis, University of Washington, Washington, USA, 167 p.

Niles, J.O. and Schwarze, R. (2001). The value of careful carbon accounting in wood products. Editorial. *Climatic Change 49*:371–376.

Oliver, C.D. (1992). A landscape approach: Achieving and maintaining biodiversity and economic productivity. *Jouranl of Forestry 90*:20–25.

Perez-Garcia, J., Lippke, B., Comnick, J. and Manriquez, C. (2005a). An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. *Wood and Fiber Science 37*:140–148.

Perez-Garcia, J., Lippke, B., Briggs, D., Wilson, J.B., Bowyer, J. and Meil, J. (2005b). The environmental performance of renewable building materials in the context of residential construction. *Wood and Fiber Science 37*:3–17.

Petersen, A.K., Gobakken, T., Hoen, H.F. and Solberg, B. (2004). Avoided greenhouse gas emissions when forest products substitute competing materials - effect on carbon account and optimal forest management. A case study of Hedmark County in Norway. *Scandinavian Forest Economics* 40:113.

Petersen, A.K., Gobakken, T., Hoen, H.F. and Solberg, B. (2005). Increasing the carbon benefit from a forest area – optimal forest management and cost-effectiveness. A case-study from Hedmark County in Norway. In: Petersen-Raymer, A.K. (ed.). Modelling and analysing climate gas impacts of forest management. Ph.D. thesis 2005:11, Paper V. Norwegian University of Life Sciences, Department of Ecology and Natural Resource Management, Norway.

Pingoud, K. and Lehtilä, A. (2002). Fossil carbon emissions associated with carbon flows of wood products. *Mitigation and Adaptation Strategies for Global Change 7*(1):63–83.

Pingoud, K., Pohjola, J., Valsta, L. and Karttunen, K. (2006). Tapaustarkastelu: metsien ja puutuotteiden yhdistetty vaikutus, pp. 17–29. In: Valsta, L., Ahtikoski, A., Horne, P., Karttunen, K., Kokko, K., Melkas, E., Mononen, J., Pingoud, K., Pohjola, J., Uusivuori, J. (eds.). Puu ilmastonmuutoksen hillitsijänä. Loppuraportti. Helsingin yliopisto, Metsäekonomian laitos. ISBN 952-10-3022-4. Yliopistopaino, Helsinki, 57 p.

Valsta, L., Sedjo, R.A., Pingoud, K. and Gustavsson, L. (2005). Forests and forest products in climate change mitigation. *International Forestry Review 7*(5):72.

Biomass Forest in Sweden and Carbon Emissions Balance

T. Johansson*

1 Introduction

Biofuel has an important role to play in reducing the levels of greenhouse gases. Forestry can help reduce CO_2 levels in several ways including: storing carbon in biomass, soil and wood products; substituting fossil fuel by biofuel; and substituting energy-intensive materials like cement, steel and plastics with wood products (Schlamadinger & Marland, 1996; Ericsson, 2003). In the Kyoto Protocol, forestry and forest management can be accounted for in the attempt to reduce the greenhouse gases (Kyoto Protocol, 1997). Participating countries may choose to apply changes in carbon stocks due to afforestation, deforestation and reforestation according to Article 3.3 in the protocol, and in Article 3.4, forest management activities that results in a change of the carbon stocks could be accounted for. Besides anthropogenic CO_2 emissions originating from the combustion of fossil fuels, the land-use changes could also have a considerable impact on the amount of greenhouses gases in the atmosphere. The emissions from land-use changes could be decreased by increasing the afforestation or decreasing the deforestation. Abandoned farmland could be used for establishing new forest stands.

1.1 Biofuel Use in Sweden

For centuries trees have been used in household as firewood or for charcoal production. Conventional forest management combined with biofuel utilisation has been practised during the last 40–50 years in Sweden. However the biofuel harvesting today is mainly based on large areas for economical reasons, which means that tops and branches are harvested on clear cut areas. Today special equipments for biofuel harvests are used and the market is prepared for energy production of biofuel in

---

*Professor at Department of Bioenergy, Swedish University of Agricultural Sciences, PO Box 7061 SE 750 07 Uppsala, Sweden, tord.johansson@bioenergi.slu.se

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

district heating plants. The infrastructure is well established and there is an increasing market for consumption of energy originating from biofuels. Most part of the fuelwood harvests is distributed to the district heating plants close to cities. Other parts of the fuelwood are used as firewood, wood chips and pellets for one or two-household dwellings.

The total Swedish energy supply in 2006 was 630,000 watt-hours (Wh). Biofuels provides 18% of that total, Fig. 1. Although the fuel systems in Sweden are based on oil and electric power today, there has been a dramatic increase in the use of fuelwood during the last decade. The supply of energy by fuelwood was 6% by total energy supply in 2006 (Anon, 2006). Only tops and branches after clear cutting is harvested today, but in the future small trees from cleaning and thinning operations and stumps on clear cut areas are interesting products, which could increase the total fuelwood production by 5 and 5,000–10,000 Wh per year respectively. By more efficient methods of the harvesting the production could increase by 50% or 5,000–10,000 Wh per year.

Short rotation plantations of *Salix* have been practised during the last 30 years. The planted area, 15,000 ha, has not increased during the last 20 years. Depending on the price of cereals and the political view of the importance of Bioenergy, the farmer might plant fast growing species on farmland. According to the politicians one way to reduce CO_2 emissions is to make short rotation plantations with *Salix* and other fast growing broad-leaved species on large areas of farmland.

The stump is an interesting fuelwood source and forest companies are just proving the most effective way to harvest the stumps with or without roots. Among the two most frequent tree species in Sweden, Norway spruce (*Picea abies* (L.) Karst.) and Scots pine (*Pinus sylvestris* L.), spruce is the most useable species for stump removal as the root system is flack compared with that of pine. Several research programmes have just started studying ecological effects of stump harvesting and the way to store the stumps before burning them. However in 1970s research and practical harvests

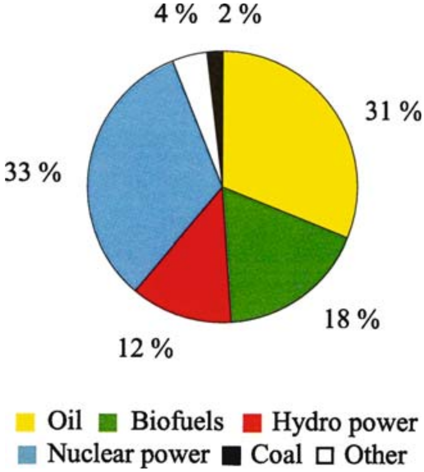


Fig. 1 Total energy supply in Sweden in 2006 (Anon, 2006)

was made but focussed on raw material for the pulp industry. Experiences from that period can be implemented today in the efforts to harvest stumps for biofuel.

Among Swedes the attitude is positive against management of forest and former farmland for biofuel production. The authorities have made rules and practical recommendations for where the tops and the branches could be harvested without risks for nutrient loss. A controversial point is plantation on farmland. Both the farmer and the public opinion want to maintain an open landscape. Another aspect is the awareness of the efforts to decrease the use of oil and, in the long perspective, nuclear power is clear and biomass utilisation is one of the possibilities to solve the energy supply by ecologically acceptable ways.

1.2 Forest Management May Affect the Amount of Carbon Emissions

Forest and forest management are an important factor in the attempt to reduce the CO_2 in the atmosphere (Kurz et al., 2002) particularly the northern hemisphere is important in the aspects of carbon storage and carbon sequestration (Liski et al., 2000; Myeni et al., 2001). Some forestry activities that can affect the amount CO_2 in the atmosphere is (Schlamadinger & Marland, 1996):

- Increase the carbon sequestration in biomass and soil
- Increase the amount of carbon stored in wood products
- Substitute fossil fuel by biofuel
- Substitute energy-intensive materials with wood products

The amount of carbon stored in wood products has not been studied to such an extent, which could be because the wood product pool currently is assumed to not change over time and thereby will not be accounted for in the Kyoto Protocol (IPCC, 2000). Forest management strategies could be an important tool in the progress to reduce the greenhouse gases. The selection of rotation length is considered to be an important and an affective strategy to affect the carbon stocks in the forest (Liski et al., 2001; Harmon & Marks, 2002). An increased rotation length results in an increased carbon stock in biomass but it is not certain that the carbon stock in the soil increases (Liski et al., 2001; Ericsson, 2003; Kaipainen et al., 2004). However, an increased rotation length might decrease the potential for logging residues to substitute fossil fuel (Ericsson, 2003). Other management strategies that could be of importance in decreasing the atmospheric CO_2 are: thinning operations, increasing the forest area, fertilization, selection of tree species. The unmanaged forest could store more carbon in the biomass than the managed forest (Cooper, 1983; Thornley & Cannell, 2000; Maclaren, 2000; Kirschbaum, 2003), which suggests that if the purpose is to store as much carbon as possible in the forest there should be no thinning operations.

When afforesting abandoned farmland carbon is removed from the atmosphere until the stand being saturated. When saturation is obtained the stand can act as a reservoir since it is keeping the carbon out of the atmosphere. New plantations have to be established annually to postpone the increase of CO_2 in the atmosphere by afforestation. But in a long-term perspective, a managed forest stand is not sequestering carbon since all biomass that is built-up during the rotation period is harvested. To increase the amount of carbon in biomass, on a regional or national level, the amount of forest land has to be increased or the forest management has to improved (Kyoto Protocol, 1997). A change of the rotation period could be an important forest management strategy to reduce the atmospheric CO_2 since different rotation period affects the capacity of storing carbon in biomass and soil (Kaipainen et al., 2004). Species with a longer rotation period have a larger average carbon stock compared with species with a shorter rotation period and could therefore keep more carbon in the biomass Maclaren (2000).

Afforesting broadleaved trees on former farmland could also be one way to increase the amount of biofuel that can replace fossil fuel. Today, most of the biofuel harvested in Sweden consists of logging residues from final cuttings in coniferous stands. These residues will probably be the main source of biofuel from the forest for a long time, but to further increase the amount other possibilities for producing biofuel must be explored. Using residues from thinning operations can be one solution. But the growth of the remaining trees could decrease when the residues are removed (Jacobson et al., 2000). It can be a suitable strategy to use trees, growing on abandoned farmland, for biofuel as it gives considerable amounts of biofuel (Johansson, 2000b). The distance to the power plants are often short (Johansson, 1999a). The amount of biofuel that can substitute fossil fuel is affected by the rotation period (Ericsson, 2003). However, managing the stands for biofuel production affects the capacity for storing carbon in biomass and soil (Wihersaari, 2005). It is not possible managing forest stands for both maximum carbon storage and biofuel production (Kirschbaum, 2003).

Paul et al. (2002) discussed that trees established on abandoned farmland could be managed on a short rotation period (i.e. 10–15 years) for energy or pulpwood with no intermediate thinning or on a long rotation period (20–50 years) for timber or veneer but production of timber requires several thinning operations and longer rotation periods. An alternative of producing timber could be to manage the trees on former arable land for a long rotation period but without any thinning operations. The amount of wood produced on these dense stands could be used for energy. A longer rotation period could also result in a larger carbon stock in the biomass (Kaipainen et al., 2004).

2 Biomass Production

Below three models for biomass production is described. The chosen models are based on results from earlier studies. In practise these models are easy to use, especially on fertile soils with broadleaf species growing in dense stands.

Table 1 Characteristics of young and mature stands of European aspen (Populus tremula L.), silver birch (Betula pendula Roth), Downy birch (Betula pubescens Ehrh.), common alder (Alnus glutinosa Gærtn.) and grey alder (Alnus incana Moench) (After Johansson, 1999a, c, d, 2000a, 2002)

Tree species	No. of stands	Age (years)	Stems (ha^{-1})	MAI (tonnes ha^{-1} year^{-1})
Young stands				
European aspen	11	5–24	6,800–46,150	2.86–9.15
Silver birch	10	7–32	2,280–45,500	0.76–8.44
Downy birch	10	6–20	2,737–298,000	0.39–5.07
Common alder	10	4–36	1,325–40,000	0.50–7.71
Grey alder	15	5–35	3,000–94,000	0.66–9.86
Mature stands				
European aspen	38	26–91	245–3,866	1.23–7.00
Birch				
Common alder	32	21–91	431–2,974	1.48–6.06
Grey alder	26	21–66	546–4,031	2.07–5.51

Fig. 2 Cutting fuel wood in a stand of ingrowths of aspen and birch (left) and the harvested fuel wood (right)

2.1 Ingrowths on Former Farmland

When farmland areas are laid down as a result of too high agricultural production combined with the lack of young people interested in farming a lot of areas are not cultivated in any way. According to Johansson (1999b), 348,000 ha of farmland was abandoned in Sweden between 1974 and 1999, and 231,100 ha of this land has not been used for any purposes such as planting trees or other forestry activities.

In a study of ingrowth on former farmland of young and mature broadleaf stands the biomass production was estimated (Johansson, 1999c, d, 2000a). The young stands were <15 years old. In the studied single stand one broadleaf species was the main individual, e.g. downy or silver birch (*Betula pubescens* Ehrh. and *Betula pendula* Roth respectively) or European aspen (*Populus tremula* L.) or grey or common alder (*Alnus incana* (L.) Moench and *Alnus glutinosa* (L.) Gærtn.). The mature stands of the same species were between 20 and 90 years old (Johansson, 1999a, 2002). Some characteristics for the studied species are given in Table 1.

The stand could be managed in a conventional way by cleanings, thinnings and clear cut but also for biofuel usage. In the latter management all of the standing biomass is removed at 15 to 20 years of age or most of the broadleaves are removed, Fig. 2.

2.2 Short Rotations of Norway Spruces Planted on Farmland

When abandoned farmland areas are planted with conifers they will grow very fast. The problem for forest industry will be a low basic density with smaller amounts of fibre per cubic meter than for more slowly grown trees on forest land. The sawn products can not be used for constructing purposes. Therefore one solution could be to manage the stand without cleanings and thinnings until the selfthinning phase starts and clear cutting (Johansson, 1999e).

2.3 Mixed Forest of Broadleaves and Conifers

A third alternative is mixed forest mostly a conifer and a broadleaf specie. The broadleaves grow faster than the conifers and must be cut earlier. The idea with a managed mixed stand is to utilize the different ways of growing pattern both in the ground (e.g. root system) and the above-ground part. Studies in mixed stands show that the growth of conifers is not disturbed if the broadleaves are cut before competition, which means not later than a 40-year-rotation of the broadleaves. The figures are from studies in Nordic countries (Braathe, 1988; Johansson, 2000b).

3 Biomass and the Carbon Emission

Below some calculations on biomass production and carbon stock are presented together with examples of methods for management of the stands. Data and management methods are taken from Swedish conditions but the conclusions are general only depending on species and geographical localisation. The influence on carbon stock is discussed.

3.1 Ingrowths on Former Farmland

Data to this overview has been taken from a study by Eriksson and Johansson (2006) of young and mature stands and calculations of the effects on CO_2 storage. Based on the estimated yield of broadleaf species growing in dense stands on fertile sites some calculations about the relationship between different managements of biomass forests and CO_2 are made. Especially the difference between short and long rotation periods is studied. The influence on management for stands with a low mean annual increment (MAI) is discussed. Characteristics for the studied stands are given below in Table 2.

When managing the young stands there is no conventional way to obtain the potential growth capacity. As shown in Table 2 MAI for a specie differ between

Table 2 Characteristics of young and mature stands of European aspen (Populus tremula L.), silver birch (Betula pendula Roth), downy birch (Betula pubescens Ehrh.), common alder (Alnus glutinosa Gærtn.) and grey alder (Alnus incana Moench) (After Eriksson and Johansson, 2006)

Tree species	No. of stands	Age (years)	Stems (ha^{-1})	SI H40 m	MAI	Moisture (cont. %)
Young stands						
European aspen	7	10–20	5,964–16,500	–	2.9–8.6	46
Silver birch	5	10–17	3,301–45,500	–	0.9–8.3	40
Downy birch	4	11–20	2,737–32,400	–	0.5–5.4	40
Common alder	4	10–17	3,861–21,600	–	1.7–4.5	43
Grey alder	8	10–17	7,400–47,600	–	2.0–8.8	45
Mature stands						
European aspen	19	32–64	846–3,242	16–20	1.4–6.9	59
Silver birch	8	26–50	950–4,061	21–28	1.6–6.7	41
Downy birch	9	30–57	838–3,253	19–27	1.7–7.3	41
Common alder	14	31–61	826–2,994	15–23	2.4–5.3	53
Grey alder	12	30–66	904–4,031	17–24	1.9–4.5	54

localities depending on stem density, site conditions and other reasons (damages by frost, wild habitat, etc.).

In the examples below the rotation period for young stands was set to 15 years. After cutting, a new stand was immediately established by sprouting and/or suckering. Totally three rotation periods á 15 years each were passed. The mature stands were harvested after 45 years. The harvests in both the short and the long rotations were used for biofuel. As the MAI for young stands could vary between the rotation periods two scenarios besides the "normal" were considered. The MAI in the second and third rotations was set to 25% higher or lower than in the first rotation.

When calculating the amount of biofuel that could be produced and used for replacing fossil fuel some data must bell introduced. The effective heating value (q_v (net)) reported by Nurmi (1993) and moisture content of each species (Table 2) were used for calculating heating values for wet biomass (q_v (moist)). The energy efficiency of generating power from biofuel was set to 42% (Ekström et al., 2001). The energy content in broadleaf stands and the amounts carbon emitted as CO_2 when combusting coal with equivalent energy contents were calculated. The averaged carbon storage stock during the rotation periods was calculated to evaluate carbon storage differences between short and long rotations.

Results of replacing coal with biofuel from broadleaved stands managed with short or long rotation periods are presented in Fig. 3. After three rotation periods the young stands of European aspen, silver birch, grey alder and common alder substituted 32% to 112% more carbon than the mature stands of these species. CO_2 emissions avoided by growing downy birch were lower with short rotations than with long rotations. When the MAI was 25% lower in second and third rotations the carbon substitutes were higher for European aspen, silver birch and grey alder than for the mature stands. Biofuels from trees grown in short rotations with 25% higher MAI could replace more carbon than mature stands for all species.

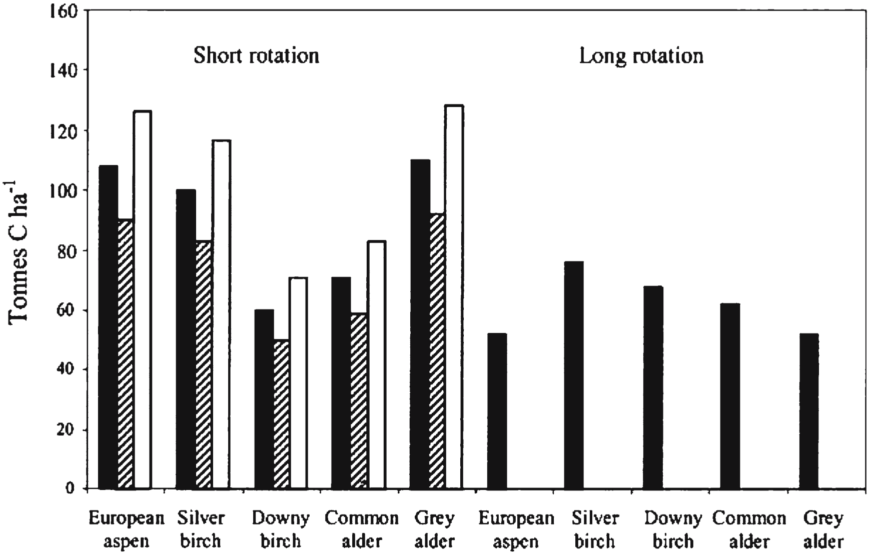


Fig. 3 Avoided immersions of CO_2 when replacing coal with biofuel after three rotation periods for the young stands and one rotation period for the mature stands. MAI for the second and third rotation period was equal in all rotations (■), decreased by 25% () and increased by 25% (), respectively, for the short rotation period

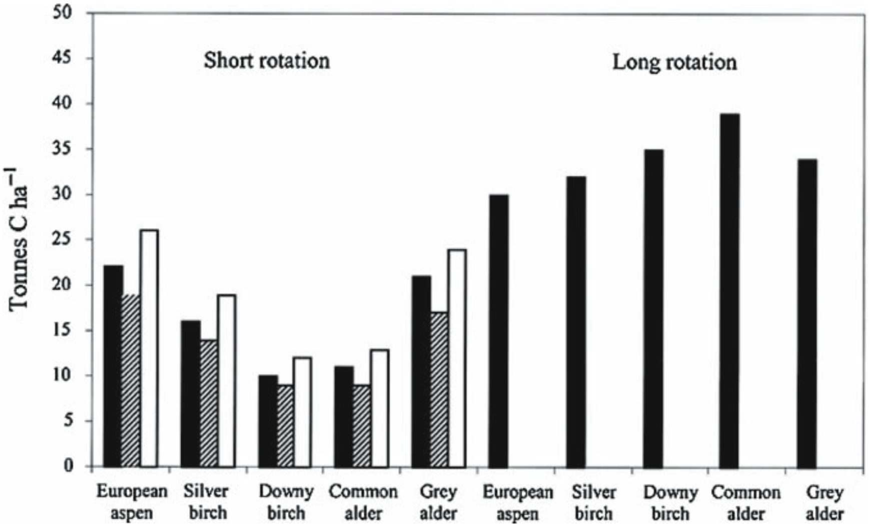


Fig. 4 Average carbon stock after three rotation periods for the young stands and one rotation period for the mature. MAI for the second and third rotation period was equal in all rotations (■), decreased by 25% () and increased by 25% (), respectively, for the short rotation period

As shown in Fig. 4 the average carbon stock varied from 32 to 41 t C ha^{-1} for the mature stands and 11–21 t C ha^{-1} for the young stands. When increasing the MAI with 25% for the second and the third rotation in the short rotation scenario the mature stands for all species stored more coal.

Table 3 Characteristics of Norway spruce stands growing on former farmland

	SI	Age	Diam.	Stem (ha^{-1})	Age	Diam.	Stem (ha^{-1})
		Before first treatment			After last treatment		
Mean ± SE	34.4 ± 0.6	29 ± 1	117 ± 4	3,330 ± 189	45 ± 2	167 ± 4	2,293 ± 75
Range	31.5–36.9	23–37	98–155	1,820–4,533	37–52	148–14	1,320–2,743

If the objective is toonaximize the average carbon stock the rotation period should be long according to the noted findings. When the MAI is low under short rotation regimes the average carbon stock will be low compared with the long rotations.

3.2 Short Rotation of Norway Spruces Planted on Farmland

When managing Norway spruce planted on former farmland the wood quality, e.g. basic density and the number of knots per tree is big. Then one way to manage this type of stand could be by no thinning and clear cut at the time for self thinning. In a study of Norway spruce planted on farmland this period will be at 40–45 years of age. To increase the biomass production of Norway spruce the most promising is to focus on the most fertile farmland areas. Below the stand characteristics have been taken from long-term experiments dealing with thinning programmes in Norway spruce and Scots pine stands. The stands have been examined every 5 to 10 years completed with thinning operations according to an experimental plan. The treatments started when the stands were 30–35 years old. One parcel (30 × 30 m) at each locality was not thinned. In the calculation ten stands were used. The studied stands had a site index, SI, of >30 m (H_{100}) and >1,500 stems per hectare when the experiment was initiated, Table 3.

The biomass production of unthinned stands during the studied period is shown in Fig. 5. The amount of biomass at 30, 40 and 50 years was 145, 193 and 204 t d.w. ha^{-1}. According to the calculation of CO_2 emissions described earlier for ingrowths of broadleaves the emission form the spruce stands were calculated. The average CO_2 emissions were 61, 81 and 103 t C ha^{-1} after 30, 40 and 50 years rotation.

3.3 Mixed Forest of Broadleaves and Conifers

Mixed forest is the most common structure of forest stands. If no management is done the different species will compete with other and make different kinds of mixture. Mostly broadleaved species especially the colonizing species, e.g. the light demanding and fast growing species as *Alnus*, *Betula* and *Populus* in Europe (Johansson, 2000b), Fig. 6.

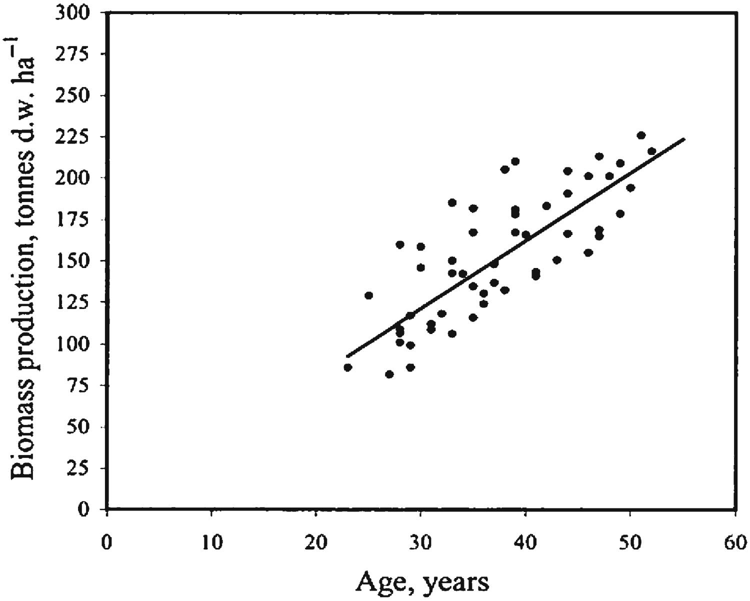


Fig. 5 Biomass production (tonnes d.w. ha^{-1}) in self-thinning Norway spruce stands

Fig. 6 Ingrowths of birch and Norway spruce before (left) and after (right) cutting of 75% of birch stem number

When the all the birches were harvested in the mixed stands after a rotation of 40 years the biomass production of Norway spruce in the shelter was 32.9 and without shelter 37.9 t d.w. ha^{-1} (Johansson, 2000b), Fig. 7. The harvested birch biomass was 41 (range 14–65) t d.w. ha^{-1}. Based on the mean value the CO_2 emissions by managing birches in a Norway spruces stand was 17 t C ha^{-1}. As the surplus of birches in the Norway spruce stand did not drastically decreased the growth of spruces, carefully managed mixed stands of broadleaves (alder, birch or aspen) and spruces both gives pulp wood or biofuel and less CO_2 emissions than if only a stand of spruces has been practised. In the conclusions of the referred study it was stated that a correct management of the mixed stands with a cleaning in the birch stand at

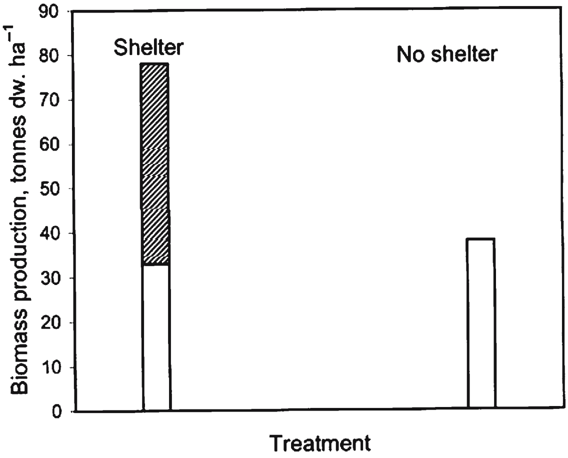


Fig. 7 Standing biomass (tonnes d. w. ha^{-1}) of Norway spruce and birch in mixed stands after a rotation of 40 years

10 years of age the biomass production should have been higher than reported (Johansson, 2000b).

4 Conclusions

In Sweden strong efforts are made to reduce the use of oil and nuclear power by utilising forest land and farmland for harvesting biofuel. By efficient methods the harvest today could be increased by 50%. Stump harvest is tested today.

Special equipments for biofuel harvests are used and the market is prepared for energy production of biofuel in district heating plants. The infrastructure is well established and there is an increasing market for consumption of energy originating from biofuels. Most part of the fuelwood harvests is distributed to the district heating plants close to cities.

As biofuel harvest in Sweden is based on large scale systems on clear cut areas (5–20 ha) the Swedish experiences could only be used as guide lines for other countries. Some modifications must be done in central and southern Europe as management systems mostly are based on small scale operations.

Three principal ways to manage forest stands for biomass production and its influence on CO_2 emissions have been presented. All results are based on stand level, e.g. the stand will be harvested and then carbon is not fixed any more. To avoid or decrease CO_2 emissions a systematic management of forest stands must be done on national and international level.

The models show that the emissions will be reduced compared with no forest or conventional forest management including cleanings and thinnings for production of pulp wood and timber.

An important factor is the soil fertility, e.g. it should be high site indices.

Another factor is management of stand with high stem density both in a short and a long rotation period.

If the objective is to maximize the average carbon stock the rotation period should be long according to the noted findings. When the MAI is low under short rotation regimes the average carbon stock will be low compared with the long rotations.

References

Anon (2006). Swedish statistical yearbook of forestry 2006. Swedish Forest Agency, Jönköping, 337 pp.

Braathe, P. (1988). Utviklingen av gjenvektst med ulike blandningsforhold mellom barrtrær och løvtrær. II. Summary: development of regenerating with different mixtures of conifers and broadleaves. II. Reports from Norwegian Forest Research Institute. Report 8, 50 pp.

Cooper, C.F. (1983). Carbon storage in managed forests. *Canadian Journal of Forest Research 13*, 155–166.

Ekström, C., Amnell, G., Anheden, M., Eidensten, L. And Kierkegaard, G. (2001). Biobränsle från skogen. En studie av miljökonsekvenser och ekonomi för olika användningar. (Biofuel from the forest. A study of environmental impact and economy during different fields of application). Report U 01:30, Stockholm, 117 pp. (in Swedish).

Ericsson, E. (2003). Carbon accumulation and fossil fuel substitution during different rotation scenarios. *Scandinavian Journal of Forest Research 18*, 269–278.

Ericsson, E. and Johansson, T. (2006). Effects of rotation period on biomass production and atmospheric CO_2 emissions from broaleaved stands growing on abondoned farmland. Silva Fennica 40(4), 603–613.

Harmon, M.E. and Marks, B. (2002). Effects of silvicultural practices on carbon stores in Douglas-fir – western hemlock forests in the Pacific Northwest, U.S.A.: results from a simulation model. *Canadian Journal of Forest Research 32*, 863–877.

IPCC Special report – land use, land-use change, and forestry. Summary for policymakers (2000). ISBN: 92-9169-114-3. Available from http://www.grida.no/climate/ipcc/spmpdf/srl-e.pdf. [Cited 10 Jan 2006].

Jacobson, S., Kukkola, M., Mälkönen, E. and Tveite, B. (2000). Impact of whole-tree harvesting and compensatory fertilization on growth of coniferous thinning stands. *Forest Ecology and Management 129*, 41–51.

Johansson, T. (1999a). Dry matter amounts and increment in 21- to 91-year-old common alder and grey alder and some practical implications. *Canadian Journal of Forest Research 29*, 1679–1690.

Johansson, T. (1999b). Förekomst av självföryngrade lövträd på nedlagd jordbruksmark. Summary: presence of self-regenerated broad-leaved trees growing on abandoned farmland. Department of Forest Management and Products, Swedish University of Agricultural Sciences, Report 2, 83 pp.

Johansson, T. (1999c). Biomass equations for determining fractions of pendula and pubescent birches growing on abandoned farmland and some practical implications. *Biomass and Bioenergy 16*, 223–238.

Johansson, T. (1999d). Biomass equations for determining fractions of European aspen growing on abandoned farmland and some practical implications. *Biomass and Bioenergy 17*, 471–480.

Johansson, T. (1999e). Biomass production of Norway spruce (*Picea abies* (L.) Karst.) growing on abandoned farmland. *Silva Fennica 33* (4), 261–280.

Johansson, T. (2000a). Biomass equations for determining fractions of common and grey alders growing on abandoned farmland and some practical implications. *Biomass and Bioenergy 18*, 147–159.

Johansson, T. (2000b). Regenerating Norway spruce under the shelter of birch on good sites might increase the biofuel supply in Sweden. *New Zealand Journal of Forest Sciences 30* (1/2), 16–28.

Johansson, T. (2002). Increment and biomass in 26- to 91-year-old European aspen and some practical implications. *Biomass and Bioenergy 23*, 245–255.

Kaipainen, T., Liski, J., Pussinen, A. and Karjalainen, T. (2004). Managing carbon sinks by changing rotation length in European forests. *Environmental Science and Policy 7*, 205–219.

Kirschbaum, M.U.F. (2003). To sink or burn? A discussion of the potential contributions of forests to greenhouse gas balances through storing carbon or providing biofuels. *Biomass and Bioenergy 24*, 297–310.

Kurz, W.A., Apps, M., Banfield, E. and Stinson, G. (2002). Forest carbon accounting at the operational scale. *The Forestry Chronicle 78* (5), 672–679.

Kyoto Protocol (1997). Kyoto Protocol to the United Nations Framework Convention on Climate Change. Available from: http://unfccc.int/resource/docs/convkp/kpeng.pdf. [Cited 10 Jan 2006].

Liski, J., Pussinen, A., Pingoud, K., Mäkipää, R. and Karjalainen, T. (2001). Which rotation length is favorable to carbon sequestration? Canadian Journal of Forest Research 31, 2004–2013.

Liski, J., Pussinen, A., Pingoud, K., Mäkipää, R. and Karjalainen, T. (2001). Which rotation length is favourable to carbon sequestration? *Canadian Journal of Forest Research 31*, 2004–2013.

Maclaren, J.P. (2000). Trees in the greenhouse – the role of forestry in mitigating the enhanced greenhouse effect, Rotorua, New Zealand. *Forest Research Bulletin* 219, 72 p.

Myeni, R.B., Dong, J., Tucker, C.J., Kaufmann, R.K., Kauppi, P.E., Liski, J., Zhou, L., Alexejev, V. and Hughes, M.K. (2001). A large carbon sink in the woody biomass of Northern forests. *Proceedings of the National Academy of Sciences of the USA 98* (26), 14784–14789.

Nurmi, J. (1993). Heating values of the above ground biomass of small-sized trees. *Acta Forestalia Fennica 236*, 30 pp.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G. and Khanna, P.K. (2002). Change in soil carbon following afforestation. *Forest Ecology and Management 168*, 241–257.

Schlamadinger, B. and Marland, G. (1996). The role of forest and bioenergy strategies in the global carbon cycle. *Biomass and Bioenergy 10*, 275–300.

Thornley, J.H.M. and Cannell, M.G.R. (2000). Managing forests for wood yield and carbon storage: a theoretical study. *Tree Physiology 20*, 477–484.

Wihersaari, M. (2005). Aspects on bioenergy as a technical measure to reduce energy related greenhouse gas emissions. Espoo, Finland, VTT Publications 564. 93 p. + app. 71 p.

Taper Equations and Wood Products: Assessing the Carbon Flow of the Forest Through Its Products

I. Lizarralde[1]*, M. Broto[1], F. Rodríguez[1], and F. Bravo[2]

1 Introduction

In recent years, from the signing of the Kyoto Protocol in 1997 until it came into force in February, 2005, thousands of publications have been released relating to the fulfilment of the protocol and ways of measuring carbon fluxes all over the world. From the point of view of forests, it was considered a great opportunity to enhance the value of our forests, beyond the traditional accounting of wood production and its value.

The sequestration accounting methodology proposed by the UNFCCC (United Nations Framework Convention on Climate Change) through the IPCC (Intergovernmental Panel on Climate Change) does not take into account forest products as a carbon sink (IPCC, 2000a), assuming that all carbon in the harvested biomass is oxidised in the removal year (Dias et al., 2005). As is known, a large part of the carbon remains stored in long-lived wood products and persists for decades (IPCC, 2000b). Furthermore, wood may indirectly reduce carbon emissions since it can be a substitute material for steel or concrete in construction (Werner et al., 2005).

Although several countries, as well as the UNFCCC, have recognized the important role products can play in global accounting, they can not be included until the second commitment period (2013–2017). Nevertheless, countries are nowadays free to include them in their own inventories.

Forest modelling techniques can be very useful tools for the accounting of carbon fixation in forests and, more precisely, in forest products. Forest models have normally been widely used for yield and growth estimation of our

---

[1] Cesefor, Soria, Spain, *ilizarraldet@cesefor.com

[2] F. Bravo, Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain fbravo@pvs.uva.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

forests, but the link with the sequestration of CO_2 has not been significantly explored.

In order to make an approach to this linkage, different examples are here described. On the one hand, a typical, well-known forest in Spain has been used: "Pinar Grande" is one of the largest public-managed forests in Soria (Northern Iberian Range, Spain) with an extension of over 12,500 ha, where Scots pine (*Pinus sylvestris* L.) occupies around 70% of the forest. Moreover, it was one of the first forests in Spain to have a management plan (it is now 100 years since the first plan). On the other hand, classical and new management paths for the species are compared, including material substitution rates and fluxes. The main aim of this chapter is thus to assess, with real data, the importance of wood products on carbon fixation.

2 Estimation of Carbon Stocks

As we have said, wood products are not included nowadays for the Kyoto protocol, but the UNFCCC (2003) already assumes different options for the inclusion of wood products in future commitment periods. The three main options are to consider the products as a separate activity, as a separate, non-site specific pool, and as a separate pool attached to eligible activities and land areas.

The estimation of carbon stocks in products requires a two-step process, including both the quantifying and the accounting of the carbon.

The quantifying step is based on the estimation of the quantity of carbon fixed in the products and the time that the carbon remains fixed, while accounting refers to the estimation of product and carbon fluxes between and within countries. Quantifying, as focused in this chapter, has a merely technical component, taking into account both the forest and the products. On the other hand, accounting has both technical and political components, due to relationships between countries, where the relationships between CO_2-emitting and CO_2-fixing countries stand out. The European Union has to decide on the best method for accounting, probably among the different already existing accounting trends. These trends are:

IPCC default approach:

This approach basically assumes there is no change in the size of the wood products pool. Emissions from harvested wood are attributed to the year of production and to the country of harvest. This method overestimates the emissions because it supposes that all the harvested wood is burned or disposed of in solid waste disposal sites, when, depending on the country, a large portion is usually converted into wood products.

Stock-change approach:

This method estimates the net changes in carbon stocks in the forest and wood-products pool. Furthermore, it offers incentives to Sustainable Forest Management policies and to the use of bioenergy and long-lived products. Changes in carbon stocks in forests are accounted for in the country where the wood is grown (the producer country) and changes in the products pool are accounted for in the country where the products are used (the consuming country). In this way, the stock-change approach benefits the consuming countries. Nowadays, this method is preferred by a majority of the countries in the European Union.

> Production approach:
>
> With the production method, net changes in carbon stocks in the forests and in the wood-products pool are estimated, but all the changes are taken into account for the producing country: that is to say, only domestically produced wood is taken into account. All the changes are computed when, but not where, they occur. This method is nowadays rejected by the majority of EU countries.
>
> Atmospheric flow approach:
>
> The idea of the atmospheric-flow approach is to account for net emissions or removals of carbon to or from the atmosphere within the national boundaries, including when and where they occur. It focuses on consumption, and so both imports and exports are taken into account. This method clearly benefits producer countries with low consumption, because the producing country only reports emissions from harvesting while the consuming country does not increase its carbon pool with the imported wood products but must report the emissions when these products decay.
>
> A new, mixed approach:
>
> Besides these official approaches, another new approach could be proposed in terms of the useful life of the different products. In this chapter we propose a new approach based on forest products evaluation. The official approaches are related to carbon stocks in products but ignore the useful lives. Thus, this proposed new approach would take into account the sequestration for consuming and the emissions for harvesting within the country of use, but weighting the value of each product according to their useful life. The sequestration of the forest is assigned to the producing country, once the CO_2 is evaluated, absorbed by the products obtained from the harvesting of that forest. Furthermore, the harvesting is not considered as an emission, but rather as a transmission to the country of use which has to account the quantity as fixation and needs to absorb the production and transportation emissions which those products have required.

The useful life can be defined by two different terms. On the one hand, “half-life” refers to the time after which half the carbon placed in use is no longer in use, assuming a destruction function for the product unit (Skog & Nicholson, 2000). On the other hand, “average lifetime” is the average time a product is in use, using a linear function of product decaying.

Thus, the life cycle of the products can be defined as the group of transformations of a product from the harvesting until its final disposal in landfills or burning, including recycling and reusing.

When assuming the carbon sequestration by products, it is not only the storage of carbon which has to be accounted. The recycling fluxes and the substitution of other products appear as a key factor for the use of products in accounting methods.

In general, substitution is defined as any use of biomass that reduces the use of the non-biomass materials. According to data from the European Union, in the case of wood products, the impact of the substitution of materials on the mitigation of climate change can be even greater than the impact of the sequestration (EU, 2004). This impact on climate change derives from the fact that producing wood products consumes much less electric or fossil energy than producing other materials such as steel or concrete. It has been demonstrated (EU, 2004) that producing a fixed quantity of concrete needs about double the energy needed to produce the same quantity of wood. In the case of steel and aluminium, this rate reaches tens of times the energy needed for producing wood. In order to calculate the effects of substitution, a simple index is used called the "substitution factor", and calculates the reduction of CO_2 emissions due to the use of wood rather than other materials. The equation for this substitution factor is:

$$S = \Delta C/\Delta P = INCREMENT\ OF\ EMISSIONS/INCREMENT\ OF\ WOOD\ USE \quad (1)$$

3 Materials and Methods

3.1 Data

The data used for this work came from the third National Forest Inventory and from inventory data measured for the last revision of the management plan of "Pinar Grande" forest, which was carried out in 1995. In the typical inventory, a summary of the number of trees and volume per diametric class is obtained. Furthermore, a cutting plan for the next ten years is proposed. This cutting plan is usually divided into: (i) regeneration cutting (in order to obtain optimal conditions for the regeneration of the stand by cutting down old trees) and (ii) thinning (for the better development of the stand). In a sub-sampling of the inventory, some trees were felled and the total height was measured. These measures are used to develop volume equations which provide information for the actual volume of the forest and for the cutting plan.

3.2 Methods

The development of the process has been designed as follows: the availability of an inside-bark taper equation developed for the species in the study location (Lizarralde & Bravo, 2005) made it possible to introduce data for the trees felled (Diameter at Breast Height and Total Height) in the cited equation. At the same time, this equation is part of Cubica (Rodríguez & Broto, 2003) and CubiFor (Rodríguez et al., 2007), a software and excel complement that resolves the equation and gives volume and product classification as outputs. In this way, assuming a mean tree for each diametric class, the product classification by diametric class is obtained. The global methodology flowchart is shown in Fig. 1.

The taper equation used for this purpose was developed for Scots pine, natural forests in Castile and León (Lizarralde & Bravo, 2005) using for its development some plots included in "Pinar Grande", so that its adaptability to the present data set is supposed to be good. The equation is the following:

$$d = (1 + 0.4159 \cdot e^{-16.733 \cdot h}) \cdot (0.7365 \cdot DBH \cdot (1-h)^{0.5869-0.8945(1-h)}) \quad (2)$$

where: d = relative diameter
h = relative height
DBH = Diameter at Breast Height

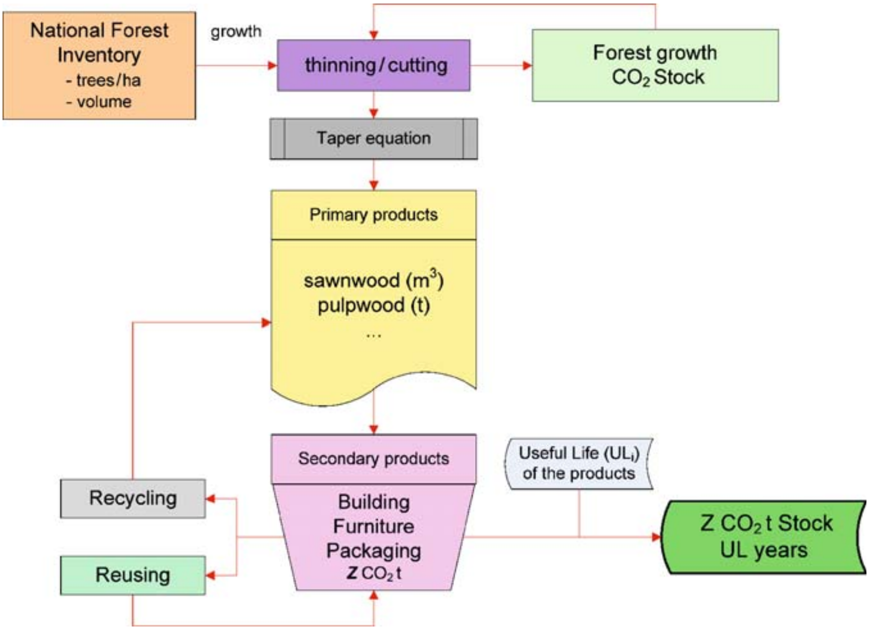


Fig. 1 Methodology for the calculation of the CO_2 stored in wood products

The cutting plan proposes a certain number of trees from each diametric class to be felled. Thus, the volume of all the trees supposed to be felled in the next ten-year period can be directly calculated with a simple product classification (pulpwood and sawn wood). Thus, based on knowledge of the sector in the region, a more realistic classification was made, obtaining commercial products through manufacturing yields of each product. Figure 2 shows the flowchart of the production of those products with the recycling fluxes.

With the addition of recycling yields and useful life data, all the wood volume per product is calculated for a ten-year period and with continuing cycles of ten years, a simulation of the carbon sequestrated is made for 50 years. Useful life calculation is based on experience of the sector, species and location from a conservative point of view, and may not agree with other assumptions (Row & Phelps, 1996; Dias et al., 2005). In Fig. 3, the useful life periods for each product are shown:

The calculation of useful lives was made via the basic useful life of each product, weighted with their importance and taking into account residues, sub-products and recycling. Thus, in order to know the time the carbon remains out of the atmosphere it is necessary to know the secondary products which are going to be produced. Useful life (ULi), reusing (using again the same product) and recycling (using the product to make new products, basically boards and paper) rates are assigned for each product. The final carbon storage is obtained by calculating a weighted useful life with the percentage of the primary product that goes to every secondary product. Equation 3 shows an example of this methodology:

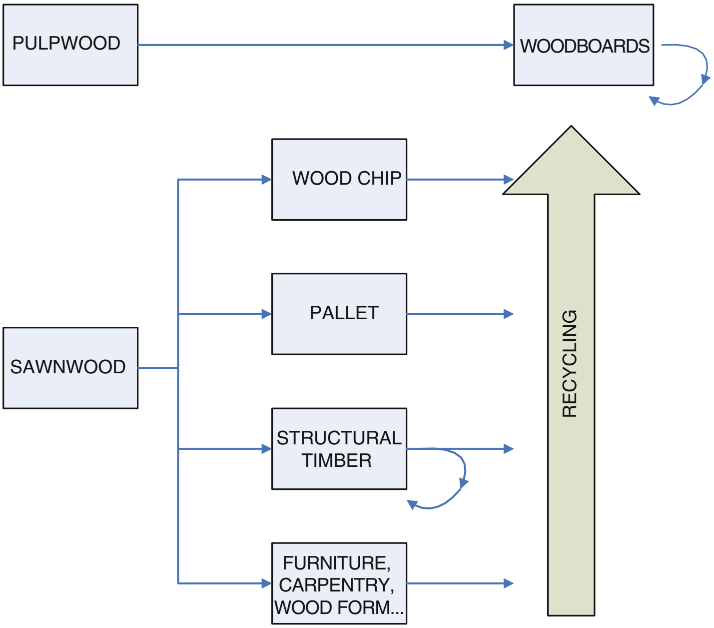


Fig. 2 Flowchart of Scots pine wood production in Castile and Leon with recycling fluxes

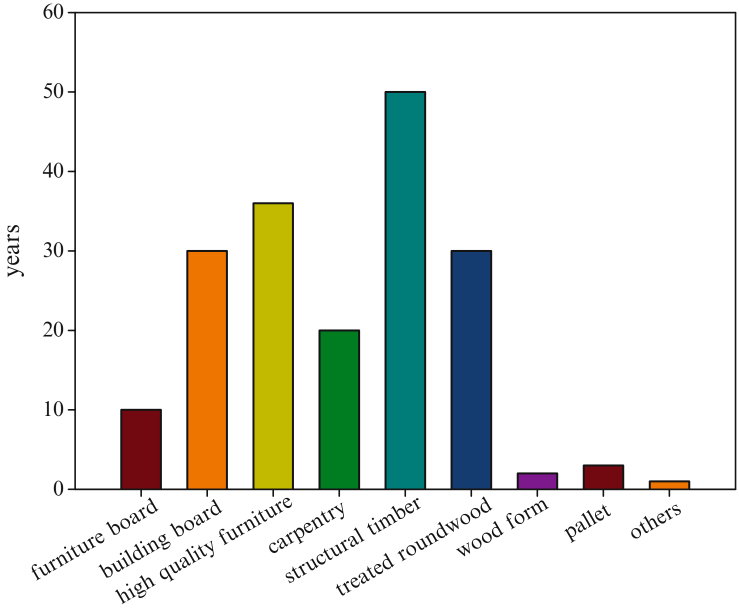


Fig. 3 Useful life of the wood products obtained in the process

$$ULi = \%j \cdot ULj + \%k \cdot ULk + \cdots + \%n \cdot ULn \quad (3)$$

Where ULi is the useful life of the primary product i

- ULn is the useful life of the different secondary products
- %n is the percentage of primary product i that goes to secondary product n
- When a product is reused or recycled, a new useful life is assigned and added to the original useful life in order to obtain the total life of the product.

4 Results and Discussion

In order to see the importance of products on the global accounting of CO_2, some examples are presented. Castile and Leon is the most important region in Spain from a forest point of view. It is estimated that the forests of the region accumulate every year about 12 million CO_2 tons. If the sequestration by the products is added to this number, the global sequestration may reach up to 15 million tons. The 3 million tons are obtained from a strong thinning simulation for a 50-year time span, where the objective is to maximize the sequestration of CO_2 based on the elaboration of products with a long, useful life.

In order to obtain a management plan which optimizes the carbon sequestration, three different management paths were simulated and a "non-management" path was included in the comparative analysis. The fixed management plans were taken from two yield tables for Scots pine in Spain. One of them is a strong thinning proposal from a yield table for the Iberian range (García-Abejon, 1981) and the other is a moderate proposal from the yield table for the species in the Guadarrama range (Rojo & Montero, 1996). The path named "Cesefor" is the one proposed in this chapter and its objective is to maximize carbon pools based on strong cutting paths. The percentage of thinning by diametric class is shown in Table 1.

With these different cutting plans, a simulation of the evolution of carbon stock (in forest and products) was done for the next 50 years. Table 2 shows the evolution of CO_2 with the different silvicultural paths.

This data produce the following figure (Fig. 4), where it can be clearly seen how forest management leads to greater sequestration, taking into account both the forest growth and the products obtained from the harvesting. The differences between the used yield tables (García-Abejón, 1981; Rojo & Montero, 1996) are not significant but the other two options are clearly different. The total balance during the first ten years is the same for all the paths because all the products are in their useful life. With the option of no cutting, growth rapidly starts to fall, and so the total balance is very low. On the other hand, the option proposed by the authors focuses on harvesting more large trees in order to obtain products with long useful lives which consequently leads to the greatest CO_2 accumulation while it avoids carbon emissions from big, old trees if not harvested.

Table 1 Thinning percentage by diametric class for Scots pine with different management paths. The "Cesefor" path is the one proposed in this chapter

DC	No cutting	Cesefor	García-Abejon (1981)	Rojo and Montero (1996)
10	0.0%	20.0%	50.3%	44.2%
15	0.0%	20.0%	38.6%	39.6%
20	0.0%	20.0%	29.7%	35.0%
25	0.0%	30.0%	22.8%	30.4%
30	0.0%	20.0%	17.5%	25.8%
35	0.0%	37.0%	13.4%	21.2%
40	0.0%	45.0%	10.3%	16.6%
45	0.0%	45.0%	7.9%	12.0%
50	0.0%	40.0%	6.1%	7.4%
55	0.0%	40.0%	4.7%	2.8%
60	0.0%	40.0%	3.6%	2.8%
65	0.0%	45.0%	3.6%	2.8%
70	0.0%	45.0%	3.6%	2.8%

Table 2 CO_2 evolution on growth, products and globally for different silvicultural paths. The "Cesefor" path is the one proposed in this chapter

	Years				
Paths	10	20	30	40	50
Growth (CO_2 tons)					
No cutting	15,977,417	16,170,680	11,192,515	5,281,162	2,161,514
Cesefor	1,110,201	762,636	54,506	−874,599	−1,275,781
G-A (1981)	7,170,277	7,697,776	8,145,751	8,557,277	8,974,851
R&M (1996)	4,643,226	1,259,296	−722,365	−1,596,384	−1,938,542
Products (CO_2 tons)					
No cutting	0	0	0	0	0
Cesefor	14,867,216	29,801,101	41,846,954	53,616,554	59,333,896
G-A (1981)	8,807,140	17,034,699	21,206,620	25,274,917	27,849,496
R&M (1996)	11,334,191	24,977,231	32,165,208	37,741,974	39,884,001
Total (CO_2 tons)					
No cutting	15,977,417	16,170,680	11,192,515	5,281,162	2,161,514
Cesefor	15,977,417	30,563,737	41,901,461	52,741,954	58,058,115
G-A (1981)	15,977,417	24,732,476	29,352,370	33,832,193	36,824,347
R&M (1996)	15,977,417	26,236,527	31,442,843	36,145,590	37,945,459

Where G-A is the yield table of García-Abejón (1981)
R&M is the yield table of Rojo and Montero (1996)

Another simple example to show the value of the forest products is to compare the carbon balance (emission-fixation) of products elaborated from different materials. The substitution of other materials for wood can be a very useful tool for decision makers to develop new strategies relating to the Kyoto Protocol. Some products have been chosen as examples of the importance of this issue. In this case, windows, building frames and utility and/or telephone poles are analysed.

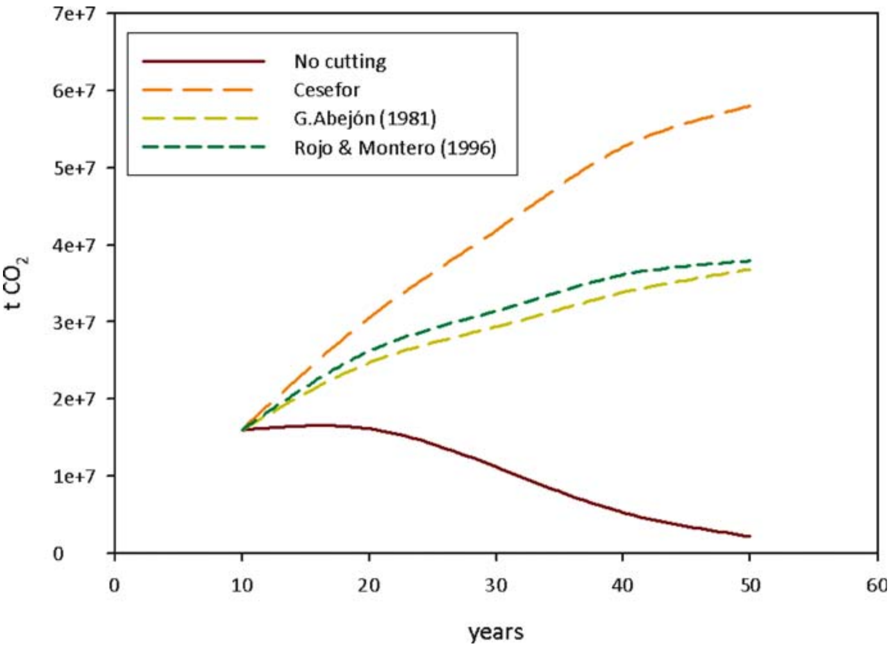


Fig. 4 Comparative analysis of CO_2 sequestration from different management paths for Scots pine in Castile and Leon

It is estimated that more than six million windows are produced every year in Spain. The balance of producing a window with wood is negative: that is to say, the carbon which the window absorbs is greater than the emissions due to the productive process of the window (about 0.32 CO_2 tons per house). Furthermore, producing a window with PVC or aluminium has a positive balance because there are only emissions and no fixation. In this way, and estimating 500,000 houses built per year, the balance of substituting, for instance, aluminium for wood would represent more than 1.8 million CO_2 tons (Lizarralde et al., 2007) which is enough to absorb the emissions of 750,000 cars.

In the case of building frames, Spain is a country with a very small tradition of building with wood compared with other countries in Europe, but substituting a concrete frame for wood in a single house means 20 t of CO_2. If this value is multiplied by the number of houses built per year, the final balance could reach about 10 million CO_2 tons. This means that with only this change, 2% of the emission rights of the country could be absorbed.

Finally, producing all the utility and telephone poles in Spain with concrete may represent the emission of more than 3.6 million CO_2 tons. Otherwise, if made with wood, 2.6 million CO_2 tons could be absorbed, that is to say, the balance is of about 6.3 million tons or an energy saving of 18,000 GWh, around 7% of the energy consumed in Spain.

Reducing the scope of the analysis (and in order to show clearly the importance of the management plans and the need to quantify carbon pools on wood products), a case study from the even-aged Scots pine "Pinar Grande" forest based on the next ten-year harvesting plan is shown.

The transformation of these volumes in products follows the usual rate of the industry for the species in this location, using most of the small-diameter wood for pulpwood which is transformed into boards and pallets, with a small part going to other products such as treated round wood. As the diameter gets larger, the assignment to sawn wood becomes bigger, although there is always a part devoted to board production. In Fig. 5, the proportion of wood assigned to each product by diametric class is shown.

Once we have simulated the cutting plan (Table 3) with the distribution of products, the sequestration of carbon by the products can be obtained. The first step in the simulation is to calculate the sequestration in the first ten-year period. Introducing the different useful lives and the recycling fluxes, a longer simulation can be done. In this case, due to the maximum useful life assumed for the products obtained, a 50-year simulation was performed.

The final result of fixation in the 8,750 ha of Scots pine in "Pinar Grande" is almost 600,000 t of CO_2 which will not be emitted into the atmosphere due to the

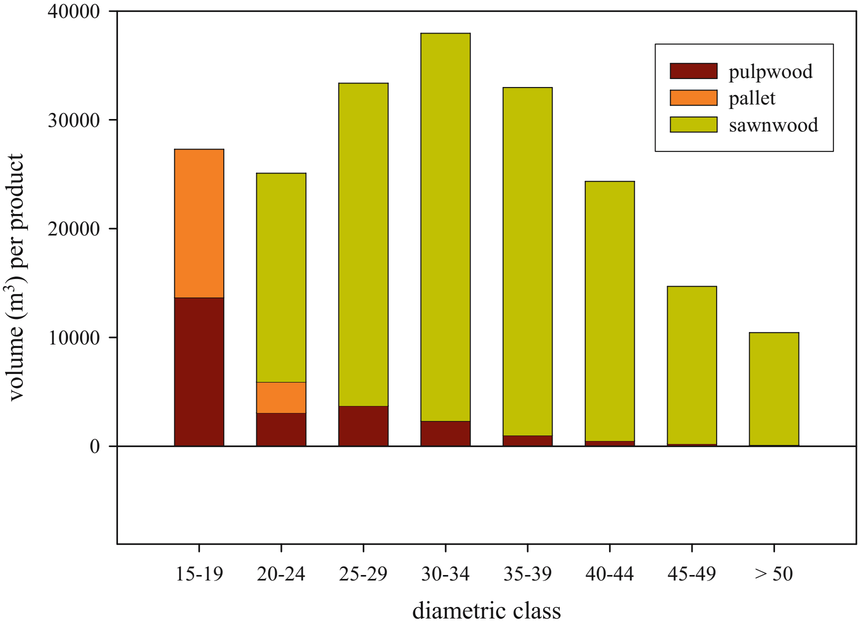


Fig. 5 Distribution of volume per product for each diametric class

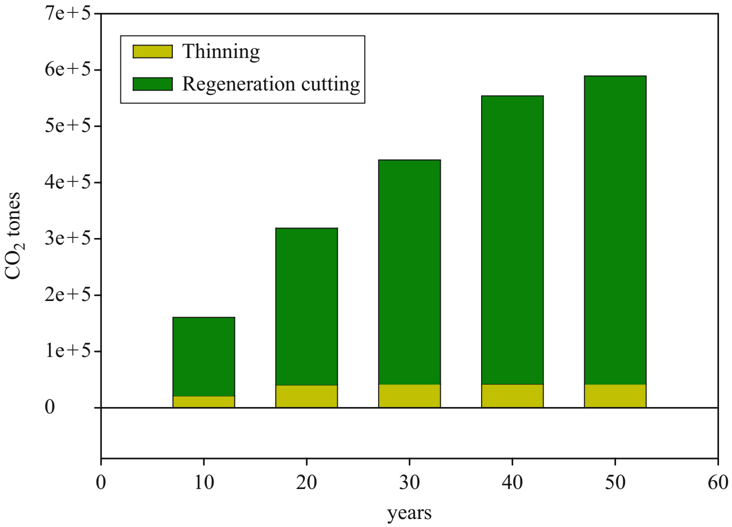


Fig. 6 CO_2 sequestration by Scots pine wood products for the next 50 years in "Pinar Grande"

Table 3 Proposed volume to be cut in the "Pinar Grande" in a ten-year period

Objective of cutting	Diametric class (cm)	Volume (m^3)
Thinning	15–19	27,300
Regeneration	20–24	25,595
	25–29	33,711
	30–34	38,339
	35–39	33,300
	40–44	24,465
	45–49	14,754
	>50	10,442

manufacturing of wood products, as shown in Fig. 6. This value does not take into account externalities such as the fixation of workers in the area or more CO_2 indirectly sequestrated by the substitution of other materials for wood. Moreover, in the period simulated only three recycling cycles are taken into account and the possible burning of the products is not observed, so that the final sequestration rate should be even greater than that shown.

The sequestration of the forest itself (taking into account only the stem of the trees) reaches some 750,000 t of CO_2. This means that the proposed harvesting level in the next 50 years will increase the sequestration capacity of the forest to 78.8% and to around 81.6% if the harvesting is concentrated on trees with a diameter larger than 35 cm.

Thinning strategies do not have a big influence on carbon sequestration in themselves, but the products which those strategies can create in the future forest will increase the sequestration rates by 7.12% if the diameters reached are around 25 cm and even by 11.5% if the diameters are bigger than 35 cm.

Recycling strategies do not have a very strong influence on the global sequestration rate. To give an example, for different products, a 20% variation in the recycling rate of a product only reflects a 2% variation in the total carbon sequestration in 50 years. It can be clearly seen that the important point is the decision as to what secondary product it is better to produce for a certain forest, rather than the recycling rates this product will have in the future.

5 Conclusions

The use of taper equations, including product classification and the assessment of different management plans, as shown in this work, may lead to a better knowledge of carbon fluxes in forests and wood products. Thus, carbon-optimizing forest policies can be developed. The methodology developed allows not only carbon accounting but also an assessment of the "quality" of that carbon: that is to say, the time the carbon will remain in the wood product depending on its use.

The integration of the wood products in global accounting of the carbon balances can represent a great advantage towards the fulfilment of the Kyoto Protocol, and is as important as forest growth. This study may be a starting point in trying to include them for the next commitment period (2013–2017). This aim requires great effort, not only from the scientists but also from governments in order to estimate the real power of our forest and their products. The use of wood products benefits us (i.e., society) from an environmental point of view. Hard work is needed to demonstrate these benefits to society and the policy makers but the final aim is worth while.

Forest management has to be the basis for the development of these kinds of policies. The choice of the management patterns and the products to be obtained is a key factor in assessing the real fixation of CO_2. Forest management determines the products obtained from harvesting. In this way, silvicultural treatment and plans focused on sustainable management and obtaining long-lived products will optimize the global influence of our forests upon climate change, and at the same time maintain our natural resources.

References

Dias, A.C., Louro, M., Arroja, L., Capela, I. (2005). The contribution of wood products to carbon sequestration in Portugal. *Annals of Forest Science, 62*, 903–909.

EU (2004). *Comprenhensive report 2002–2003 regarding the role of forest products for climate change mitigation*. http://ec.europa.eu/enterprise/forest_based/ccmreport.pdf. Last access on: 29/10/2007.

García-Abejón, J.L. (1981). *Tablas de producción de densidad variable para Pinus sylvestris L. en el Sistema Ibérico*. Comunicaciones INIA, Serie: Recursos Naturales, 29, Madrid, 36 pp.

IPCC (2000a). *Good practice guidance and uncertainty management in national gas inventories*. IPCC Technical Report Unit, Hayama.

IPCC (2000b). *Special report on land use, land-use change and forestry*. Cambridge University Press, Cambridge.

Lizarralde, I., Bravo, F. (2005). *Análisis del espesor de corteza en ecuaciones de perfil para pino Silvestre (Pinus sylvestris L.) en Castilla y León*. IV Congreso Forestal Nacional, Zaragoza.

Lizarralde, I., Broto, M., Rodríguez, F. (2007). Ciclo de vida de los productos forestales. Impacto sobre la fijación de CO_2. In: Bravo, F. (Coord.), Fundacion Gas Natural (Ed.), *El papel de los bosques españoles en la mitigación del cambio climático*. pp. 263–288.

Rodríguez, F., Broto, M. (2003). Cubica v1.2: Programa informático para cubicar y clasificar productos a partir de los datos procedentes de un inventario. *Montes, 72*, 21–29.

Rodríguez, F., Broto, M., Lizarralde, I. (2007). *CubiFor, complemento de Excel para la clasificación de productos, cubicación y cálculo de biomasa y CO_2. Cesefor Internal document*. Unedited. www.cubifor.cesefor.com. Last access on: 29/10/2007.

Rojo, A., Montero, G. (1996). *El pino silvestre en la sierra de Guadarrama*. Ministerio de Agricultura, Pesca y Alimentación, Madrid, 293 pp.

Row, C., Phelps, R.B. (1996). Wood carbon flow and storage after timber harvest. In: Sampson, R.N., Hair, D. (Eds.), *Forest and global change, volume 2: forest management opportunities for mitigating carbon emissions*. American Forest, Washington, DC, pp. 27–58.

Skog, K.E., Nicholson, G.A. (2000). Carbon sequestration in wood and paper products. In: Joyce, L.S., Birdsley, R. (Eds.), *The impact of climate change on America's forests: a technical document supporting the 2000 USDA forest service RPA assessment*. General Technical Report RMRS-GTR-59. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, pp. 79–88.

UNFCCC (2003). *Estimation, reporting and accounting of harvested wood products*. Technical paper FCCC/TP/2003/7.

Werner, F., Taverna, R., Hofer, P., Richter, K. (2005). Carbon pool and substitution effects of an increased use of wood in buildings in Switzerland: first estimates. *Annals of Forest Science, 62*, 889–902.

Forest Management Strategies and Carbon Sequestration

F. Bravo[1]*, M. del Río[2], A. Bravo-Oviedo[3], C. Del Peso[4], and G. Montero[5]

1 Introduction

Regardless of their geographical location, forests play an important role in CO_2 fixation. Carbon stored in terrestrial ecosystems is distributed among three compartments: biomass of living plants (stem, branches, foliage and roots), plant detritus (including fallen branches and cones, forest litter, tree stumps, tree tops, and logs) and soil (organic mineral humus, and surface and deep mineral soil). Trees acquire energy for their living structures through photosynthesis which requires CO_2 captured by stomata in the leaves. Part of the captured CO_2 is used to create living biomass, while the remainder is released back into the atmosphere by autotrophic respiration. When leaves or branches die, they decompose, resulting in the increase of soil carbon, although a small amount is driven into the atmosphere by means of heterotrophic respiration.

Recent climate changes have resulted in highly variable weather patterns with a general trend of increasing temperatures. These temperature increases will increase evapotranspiration in forest ecosystems. Higher CO_2 concentrations in the atmosphere will increase photosynthetic activity. Accompanying this rise in temperature is an expected drop in available water for vegetation growth. Plants will respond to reduced

---

[1] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain,* fbravo@pvs.uva.es

[2] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, delrio@pvs.uva.es

[3] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, bravo@inia.es

[4] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain, cdelpeso@pvs.uva.es

[5] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA Madrid, montero@inia.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

water availability by closing stomata, resulting in lower rates of gas exchange, reducing the positive effects of high CO_2 concentrations. In areas where water is the limiting factor for growth and survival, inter- and intra-specific competition will be more acute, especially during the regeneration. The distribution of the energy captured by plants among different functions such as stem increment, flowering, fructification, and branch and leaf formation, may be altered under erratic weather conditions that may be generated through climate change. In areas where water is the limiting factor for growth and survival, inter- and intra-specific competition will be more acute, especially during the regeneration. The distribution of the energy captured by plants among different functions such as stem increment, flowering, fructification, and branch and leaf formation, may be altered under erratic weather conditions that may be generated through climate change.

Given that forest species have an optimum range of temperatures for development, a widespread rise in the land temperature will change the competitive balance between species, and may alter their distribution patterns. It is very unlikely that the adult trees will suffer the generalised problems of sudden death, analogous to the drought affecting holm oaks (*Quercus ilex* L.), although this is possible in extreme conditions. Also, it is likely that problems in regeneration and the initial development of forest species in certain areas will occur. Fujimori (2001) pointed out that these effects will be especially serious in the marginal zones of plant distributions, where competition will be even more acute. Moreover, the problems of plagues and diseases may become very severe (Melillo et al., 1996). Conservation or even increases in biodiversity, with regards to both species and genotypes within each species, should be a management priority to reduce the effects of climate change.

Helms (1998) defined silviculture as “the art and science of controlling the establishment, growth, composition, health and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis”. The practical application of principles of diverse disciplines, including biology, ecology, and economics, to the regeneration, density control, use and conservation of forests is called forest management (Helms, 1998). Silviculture and forest management were developed as sciences in Europe in the eighteenth century to satisfy the regular and continuous necessity of providing wood for fuel and construction. Silviculture and forest management can be used in conjunction to mitigate the impacts of climate change through four fundamental strategies: (1) conserving and maintaining carbon accumulated in the forests; (2) sequestering or incrementing the carbon retained in the forests and wood products; (3) replacing fossil fuels by biomass-derived fuels; and (4) reducing the use of products that require fossil fuels in manufacturing through use of renewable forest products, such as wood, resin, and cork. The use of forests as a source of renewable energy and maintaining forest area will have the most impact worldwide.

In this section, we will describe different ways in which forests and forestry can help to mitigate climate change and the potential impact of these activities. The three compartments of carbon storage should be considered in all estimates of impacts. Carbon content in living biomass is easily assessed through species specific equations or conversion factors to estimate carbon from oven dried weight of biomass (e.g., Ibáñez et al., 2002). Litter carbon content has been analysed in many studies on primary productivity of forests, although information on how litter carbon content is influenced by forest management is not as available (Blanco et al., 2006). In the last decade, efforts have been made to assess soil carbon in forests, but studies on the effect of forest management on soils show discrepancies among studies (Lindner & Karjalainen, 2007).

2 Forest Management and Carbon Sequestration During the Last Century

In a report regarding mitigation, the Intergovernmental Panel on Climate Change (IPCC, 2001, 2007) warned of the temporality of carbon deposits in forests and of the possibility of great emissions from large-scale forest fires if not controlled. At a global scale, carbon loss due to deforestation processes, mainly in tropical zones, is offset by expanding forest area and increasing wood stocks in the temperate and boreal forests.

Human activities and land use have historically affected carbon content. For example, U.S. forests were carbon sources from 1700 to 1945. Fire suppression and forest renewal in abandoned farmland changed the trend and forests were carbon sinks until 1990 (Houghton et al., 1999). Woodbury et al. (2007) subsequent study found that total carbon stocks in U.S. had increased in the past (starting in 1990) and will continue to increase until 2010 at a similar rate to recent years. On the contrary, Pacala et al. (2001) showed a stable carbon sink in continental U.S. (excluding Alaska) with values for 1980–1989 being similar to 1990–1994. National forest plans can also change the trend in carbon storage. In China, after the social revolution in 1949, the carbon content in living biomass decreased, because of human pressure on forest resources. From 1970 to 1998, afforestation and reforestation programs started to increase forest land and, consequently, stored carbon (Fang et al., 2001).

Positive effects of forests as carbon sinks are endangered by great disturbances. Liu et al. (2002) showed that forests of Ontario, Canada could be considered carbon sinks between 1920 and 1975, whereas, after 1975, those forests became carbon sources because of large scale disturbances, such as fires, pests and intense harvesting. Forests with a minimum of management are carbon sinks in the short term, until they accumulate biomass with increased risks of fire and pest activity (Schmid et al., 2006).

In many European countries, the management of forests was initiated during the nineteenth and twentieth centuries. Since planning activities were initiated, the decisions taken and the results obtained have been recorded. As a result, we can determine how CO_2 fixation in forestry biomass has evolved. Forest management will change to include the application of sustainable silvicultural measures to increase the quantity of fixed carbon in the forests. In the woodlands of "Monte de Valsaín" (Segovia, Central Spain), in the period from 1993 to 2003, net carbon fixation was increased by 6.28% (Montero et al., 2004). Another example can be found in the woodlands of "Pinar Llano" in Valladolid (Northern Plateau, Spain) where the amount of fixed carbon is expected to increase by 7.23% in the next ten years, given the rate of gain presented in the Management Project Review (Martin, 2005). Bravo et al. (2007) analyzed different forest areas in Spain and found that the annual CO_2 sequestration rates in temperate and Mediterranean forests ranged between 0.95% and 4.96% (Table 1). These results were obtained by comparing outcomes from the second and the third Spanish National Forest Inventories using biomass equations (Montero et al., 2005).

Osorio et al. (2006) studied the evolution of the fixed carbon dioxide in Mediterranean maritime pine forests (*Pinus pinaster* Ait.) of Almazán (Soria, Central Spain). In their study, the woodland management in Almazán was reviewed for a century, from 1899 to 1999. From the original planning documents and the successive planning revisions, the number of trees by size was obtained. This information was used, together with the biomass equations developed by Montero et al. (2005), to reconstruct the evolution of CO_2 fixation in the pine woods of Almazán. During the century studied (Fig. 1), an increase in CO_2 sequestration took place, oscillating between 0.78 and 3.11 Tn/ha/year. The only exception was the period immediately prior to the Spanish Civil War (1936 to 1939), when a greater pressure on natural resources due to poverty resulted in a decrease in the fixed CO_2 in the biomass (a reduction of 1.49 Tn of CO_2per hectare per year). The levels of CO_2 sequestration in forestry biomass of these pinewoods did not recover until 15 years after the end of the war. Forest management under sustainable yield paradigm while maintaining and increasing other forest values and services, as traditionally is implemented in Almazán forests, is the best way in the long term to maintain and increase forest carbon stocks while producing products and values according with last IPCC (2007) report. The increase in carbon sequestration of the Almazán forest is in agreement with Nabuurs et al. (2003) which showed an increasing carbon sink between 1950 and 1999, based on simulated carbon storage in the European forest sector when Net Biomass Production (i.e., final production including harvest and disturbances) was used as the basis for carbon calculations.

Table 1 Annual CO_2 sequestration rates in temperate and Mediterranean forests in Spain (Adapted from Bravo et al., 2007). IFN2 and IFN3 mean second and third Spanish national inventory respectively

Zone	Years	IFN2 (10^3 Tn)	IFN3 (10^3 Tn)	Annual rate (%)
Cantabric range	1991–2000	41,696	45,433	0.9583
Pyrenees	1994–2003	120,159	185,812	4.9628
Central range	1991–2002	57,164	72,404	3.2069
Toledo mountains	1992–2001	9,619	11,802	2.2986
Sierra Morena	1994–2001	15,518	19,542	2.3324

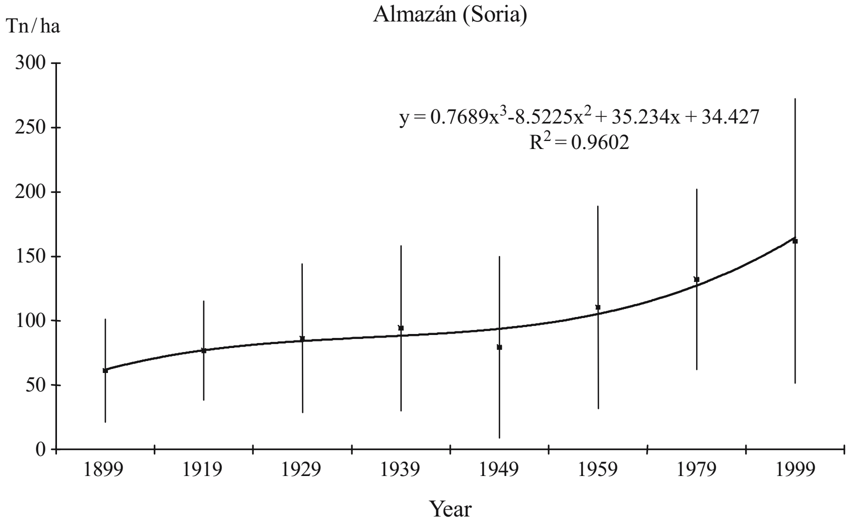


Fig. 1 Evolution of the CO_2 sequestration in the pinewoods (*Pinus pinaster* Ait.) of Almazán (Soria, Northern Spain) during the twentieth century

Nabuurs et al. (2007) stated that, in the long term, sustainable forest management strategies designed to increase or maintain forest carbon stocks while producing a constant annual yield of products (e.g., wood, fibre, etc.) and environmental services (e.g., water, biodiversity, etc.) will generate the largest sustained mitigation benefit.

3 Strategies to Improve Carbon Sequestration

The storage of carbon in forests and forest products has been proposed as an appropriate strategy for mitigating the effects of climate change. However, forest products were excluded from Kyoto protocol. To a certain extent, the use of carbon storage in forests buys time, until more definitive solutions to our dependency on fossil fuel energy are found. However, forests may become carbon sources rather than carbon sinks (Kurz & Apps, 1999; Gracia et al., 2001; Reichstein et al., 2002), since changes in regimes of such disturbances as fire, pests or drought (Fuhrer et al., 2006; Sohngen et al., 2005; Ciais et al., 2005), would affect major forest functions, forestry outputs and forest stability.

The accumulation of biomass and carbon in forest stands may be increased through different management options (Gracia et al., 2005). As examples, we can cite, among others, fire protection, pest control, increasing the length of time to rotation (harvest), regulation of tree densities, improvement of the nutritional state, selection of species and genotypes, use of biotechnology, and/or changes to the management of residues following felling. The majority of these activities are expected to increase the carbon accumulation rate by 0.3 to 0.7 Mg of C per hectare per year (Gracia et al., 2005). Management practices that alter species composition, rotation lengths, and thinning regimes, or that result in forest conservation, increases in forest land, and soil conservation can be used to increase carbon sequestration in forests.

3.1 Species Composition

Carbon storage depends on the species composition of the stand and on site quality (Bravo et al., 2008). Thus, it is known that conifer biomass contains a greater proportion of carbon per unit of biomass than broadleaved trees (Ibáñez et al., 2002). In Mediterranean areas, Scots pine (*Pinus sylvestris*) stands store more CO_2 than oak pure stands (Bogino et al., 2006), while mixed oak-pine stands have an intermediate level. Analyzing carbon soil content, in beech (*Fagus sylvatica*) forests and *Pinus nigra laricio* plantations in Calabria (Southern Italy), Scarascia-Mugnozza et al. (2001) found that beech stands store 1.47 times more soil carbon than pine plantations. Differences between pine species have been also found (Bravo et al., 2008). Co-existence with species that fix nitrogen can also increase carbon accumulation. Chiti et al. (2003) found that oak-alder (*Quercus robur* L.-*Alnus cordata* Desf.) mixed stands stored 1.18 times more carbon in the soil than pure oak stands in Tuscany (Italy), probably due to a higher humification rate. The differences found for forests with a more diverse species composition may be due to three principal reasons: (1) differences in the configuration of the stems and the crowns of tree between species; (2) differences in forest structure (e.g., the diameter distribution) generated by the different silvicultural treatments applied to mixed stands; and (3) changes in root biomass, a component whose importance is generally underestimated.

The selection of the best species composition for a stand depends on many factors (e.g., management objectives, site characteristics, etc.), but CO_2 storage should also be considered as one objective. Silvicultural treatments can be used to alter the species composition of stands, mainly through selection of species for regeneration, and through manipulation of species composition through thinning and other stand tending treatments in established plantations.

3.2 Rotation Length

During immature and mature stages of stand development, forests are carbon sinks. At older ages, forest carbon sequestration decreases slightly, or may continue to increase slowly. In order to maximize or maintain forest carbon sinks while obtaining other good and services, the length to harvest (rotation length) can be extended. Different criteria can be used to determine the appropriate rotation to simultaneously obtain forest products and achieve forest sustainability. One criterion of widespread use in the management of regulated forests is to set the rotation length at "biological rotation", when the mean annual increment (MAI) of volume per unit area is maximized. This practice maximizes wood production over the longer term (i.e., several rotations), and promotes other products and services which society demands (e.g., wild mushrooms, hunting, the conservation of the ecosystems, etc.). Rotation has an ambivalent impact on the storage of carbon in the forests (Table 2). Under rotations longer than the biological rotation, the proportion of carbon in the final harvest relative to intermediate harvests is greater (Bravo et al., 2008). Since products from harvests are often destined for long-term uses (e.g., furniture, construction, etc.), products made from these harvests after long rotations result in greater carbon storage.

If the rotation length is very long, tree mortality will occur. This results in an increase in structural diversity with dead and fallen trees, and is related to an overall increase in species diversity (Franklin et al., 1997). The decomposition rates of the dead woody materials varies with the species, size of the dead material, the type of material (i.e., bark, sapwood or heartwood), and site conditions (i.e., temperature, humidity, etc.). Dead wood has an important impact on the storage of carbon in the forestry systems because it may increase the risk of perturbations, resulting in sudden outbreaks of fires, pests, and pathogens. Other impacts to the amount of soil carbon may also occur because of the rotation length. Kaipainen et al. (2004) found a decrease in soil carbon stocks when rotation length was increased for some study cases in Europe using the CO2FIX model, indicating that soil carbon must be measured.

The biomass distribution among different components might allow different uses of the trees and, therefore, differences in the lengths of time for carbon storage. Bravo et al. (2008) found that the proportion of biomass allocated to *Pinus sylvestris* and *Pinus pinaster* stems increases with age, while decreasing in branches. Besides the importance of larger stems to carbon stored in stands, as well as in wood products created from these stems, the biomass distribution in trees obtained with longer rotations has a considerable impact on the possible use of harvest debris to generate energy. In the case of pinewoods, it has been demonstrated that the percentage of biomass for branches between 2 and 7 cm decreases with age (Bravo et al., 2008).

The proportion of carbon stocks in the final harvest, relative to the total fixed carbon in the stand, is always higher for longer rotations. However, a short rotation system produces higher carbon MAI values, regardless of the site index (Bravo et al., 2008, Table 2). Longer rotations on poor sites allow the storage of a similar amount of carbon to short rotations on good sites for *P. sylvestris*. In addition, a short rotation on poor sites does not provide stems of sufficient size to obtain a profitable income. Thus, long rotations on the poorest sites should be utilized in order to achieve both objectives: carbon sequestration and acquisition of more valuable and durable timber products. Bravo and Díaz-Balteiro (2004) showed that more extensive management systems, which involve lengthening the rotations, does result in a loss of the economic return over traditional management with shorter rotations. However, when carbon sequestration income is included in the analysis, long rotation alternatives show a positive land expectation value. Increasing harvest rotation

Table 2 The impact of species, site quality and the rotation on carbon sequestration in stands of Scots pine (*Pinus sylvestris* L.) and Mediterranean maritime pine (*Pinus pinaster* Ait.) (Adapted from Bravo et al., 2008)

Species	Site index[a]	Rotation (years)	Mean annual carbon growth (MAI) (t year^{-1})	Carbon final harvest (%)
Pinus sylvestris L.	17	83	2.16	54.61
		137	1.47	59.60
	23	69	2.99	68.12
		122	2.42	77.66
Pinus pinaster Ait.	15	101	1.28	75.19
		149	1.06	79.72
	21	83	1.89	71.91
		128	1.57	78.06

[a] Site index is the dominant height in meters at 100 years (*Pinus sylvestris*) or 80 years (*Pinus pinaster*)

length would lead to reduced harvest rate over a landscape. Under this situation some carbon pools will increase (e.g, carbon in standing trees) while others decrease (e.g., carbon in wood products) (Kurz et al., 1998). Therefore, the carbon pools dynamics at broad temporal and spatial scale should be analyzed.

3.3 Thinning

Thinning is one of the most important silvicultural interventions. Among its economic and silvicultural objectives, thinning can be used (Río, 1999):

- "To reduce competition in order to procure biological stability and improve the state of health.
- To regulate or maintain the specific composition and to prepare the stands for natural regeneration.
- To obtain production at early stages, in such a way as to maximise the production at the end of the rotation.
- To increase the value and dimensions of the products."
- The thinning regime can be defined by the age at initiation, the type of thinning, the percentage of tree stems or of basal area harvested, with respect to the values before the thinning, and the number of interventions.

Río et al. (2008) showed different thinning scenarios in Mediterranean pine stands and found that any thinning regime fixed more carbon than unthinned scenarios. However, different results were obtained when thinning regimes are compared. Balboa-Murias et al. (2006) and Pohjola and Valsta (2006) found a positive carbon sequestration if low intensities are applied, whereas Schmid et al. (2006) found that forest biomass and soil carbon increased with intense management, provided that slash was not removed and the resultant open canopy did not increase radiation and soil temperature (Paul et al., 2002). Overall, thinning is a key silvicultural treatment that affects the dimensions and wood quality of trees in intermediate and final harvests, thereby impacting the life-time of wood products.

3.4 Conservation of Forests

Although forestry activities have different carbon mitigation potentials depending also upon ecosystem features, the short-term carbon mitigation benefits through conserving current forests by reducing deforestation are greater than benefits obtained by increasing the area of forests (IPCC, 2007). Between 1990 and 2005, 13 million hectares per year of forest land were lost to other uses (FAO, 2006). South America, East Asia and Africa are the regions where deforestation rates were highest (Table 3). However, forest loss rates are decreasing around the world, except in South America (FAO, 2006). Between 2000 and 2005, Brazil (3 million hectares per year), Indonesia (1.8 million hectares per year) and Sudan (0.6 million hectares per year) suffered the largest deforestation (FAO, 2006). Conserving forests by reducing deforestation and degradation, mainly in areas where they are threatened as in the tropics, is the most effective of short-term strategies for carbon stocks preservation. Effective forest protection will lead to carbon sequestration while adaptive management of protected areas also leads to biodiversity conservation and reduced vulnerability to climate change (Nabuurs et al., 2007).

Table 3 Forest biomass carbon by regions between 1990 and 2005 and forest area in 2005 (Adapted from FAO, 2006)

	Biomass carbon (Gigatons)			Forest area (1,000 ha)	% of total land area
Region	1990	2000	2005	2005	
East and South Africa	15.9	14.8	14.4	226534	27.8
North Africa	3.8	3.5	3.4	131,048	8.6
West and Central Africa	46.0	43.9	43.1	277,829	44.1
East Asia	7.2	8.4	9.1	244,862	21.3
South and Southeast Asia	32.3	25.5	21.8	283,127	33.4
West and Central Asia	1.6	1.7	1.7	475,88	4.0
Europe	42.0	43.1	43.9	1,001,394	44.3
Caribe	0.4	0.5	0.6	5,974	26.1
Central America	3.4	2.9	2.7	22,411	43.9
North America	37.2	38.5	39.2	677,464	32.7
South America	97.7	94.2	91.5	831,540	47.7
Oceania	11.6	11.4	11.4	206,254	24.3
Global	299.2	288.6	282.7	3,952,025	30.3

3.5 Increasing Forest Area

During the last decades, reforestation of many of the marginal lands in temporate zones has increased through natural and artificial reforestation of abandoned farmlands. The increase of forest area in Europe in the last quarter of the twentieth century, prior to similar trends in the US, has lead to increased carbon reserves in living biomass as well as in the soil (Liski et al., 2002). Between 2000 and 2005, Mediterranean countries (Spain, Portugal, Italy and Greece), Vietnam and China were the greatest contributors to increases in forest area in the world, while tropical countries were the greatest contributors to decreased forest area. China (4 million hectares per year), Spain (0.3 million hectares per year) and Vietnam (0.2 million hectares per year) dramatically increased their forest areas between 2000 and 2005 (FAO, 2006). In the same period, living biomass carbon stocks decreased in Africa, Asia and South America (Fig. 2 and Table 3) (FAO, 2006). Increases in living biomass carbon stocks also increased in other countries of Europe and North America, but remained constant in Oceania. Forests of South America and Africa are the largest carbon reservoirs; therefore, conservation of forests in these continents is crucial to mitigate climate change through forest management initiatives.

Forest plantations can be used in the Kyoto protocol in accounting for emission reductions, but only in regulated circumstances by some developed countries. Increased plantation area is the main forest activity that be used to counteract carbon emissions from fossil fuels in developed nations. Between 1990 and 2005, the area of productive plantations increased from 76.8 million hectares to 109.3 million hectares (Table 4) (FAO, 2006). China (26% of the world productive plantations area),

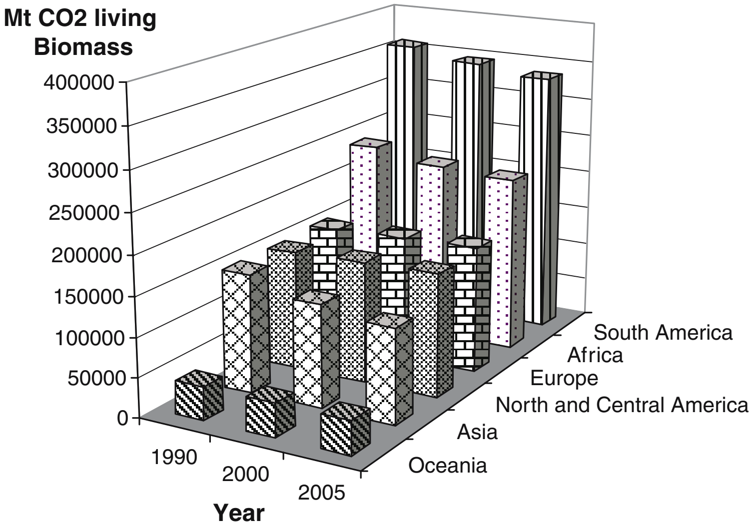


Fig. 2 Content of CO_2 (Mt) in living forest biomass by continent (Adapted from FAO, 2006)

United States (16%), Russia (11%) and Brazil (5%) are the leading countries. In that period, China increased plantation area by a factor of 1.665, or 759 thousand hectares per year. China, Russia and United States together represented 71% of new productive plantations between 1990 and 2005 (FAO, 2006). Protective plantations, for conservation purposes, increased from 20.4 million hectares in 1990 to 30.1 million hectares in 2005 (Table 4). Japan (35%) and Russia (17%) had the majority of areas planted for protective purposes (FAO, 2006).

The effect of plantations on carbon sequestration varies among the types of plantations, depending upon the plantation objectives and management, including whether the plantation is primarily meant as a productive or conservative area. Protective plantations managed for conservation (for example, through long rotation) have a limited impact on carbon sequestration, since carbon sequestration rates decline in very old plantations. In that sense, plantations for production (e.g., for wood biomass, wood for building material, etc.) represent a better carbon mitigation strategy (Lindner & Karjalainen, 2007). At each rotation, substitution with younger trees results in a net carbon emission mitigation effect.

When plantation forest projects are intended to be used to compensate for CO_2 emissions, carbon fixation for a baseline scenario (situation prior to plantation) must be compared to the expected carbon fixation output from the plantation. Also, a reliable monitoring and accounting program should be developed for lands within the project boundaries. This carbon monitoring and accounting program requires a large database and fitted biometrics models, including volume equations, biomass expansion factors, root-shoot ratios, and other previously fitted models. In some cases, this prior models or data are not locally available and substitutions must be made. Guidelines for these substitutions, in order of priority, are to use: (1) existing local and species-specific models; (2) national and species-specific models; (3) species-specific models from neighbouring countries with similar ecological

Table 4 Forest plantation area by regions between 1990 and 2005 (Adapted from FAO, 2006)

	Productive plantations (1,000 ha)			Protective plantations (1,000 ha)		
Region	1990	2000	2005	1990	2000	2005
East and South Africa	2,544	2,712	2,792	66	66	66
North Africa	6,404	6,158	6,033	1,840	2,021	2,192
West and Central Africa	1,099	1,453	1,853	70	87	112
East Asia	17,909	23,028	30,006	11,622	12,490	13,160
South and Southeast Asia	8,896	10,750	11,825	3,869	4,451	4,809
West and Central Asia	2,120	2,428	2,583	2,175	2,518	2,505
Europe	16,643	19,818	21,467	4,569	5,574	6,027
Caribe	239	243	280	155	151	170
Central America	51	183	240	32	29	34
North America	10,305	16,285	17,133	–	1,047	986
South America	8,221	10,547	11,326	10	27	31
Oceania	2,447	3,456	3,812	1	3	21
Global	76,826	97,061	109,352	24,408	28,464	30,114

conditions; or, finally (4) global species-specific models, such as those from IPCC. Uncertainties arising from these biometrics models and from sampling have to be considered in accounting. Also, CO_2 losses due to plantation activities, such as fossils fuels burned by machinery and biomass losses in site preparation prior to planting, have to be subtracted from the amount of carbon fixed. Different protocols have been approved for different plantation types and geographical areas (e.g., “Methodologies to Use Forestry as Mechanisms of Clean Devolopment, cases AR-AM0001 and AR-AM0003”[1]). These protocols must be followed to get carbon credits from forest plantation activities.

A major economic limitation to plantations as a mitigation option is the high initial investment to establish new stands coupled with the delay (usually several decades) to generate revenues (Nabuurs et al., 2007). Where forest expansion leads to a reduction of agriculture land area leading to intensive farming practices, conversion of other mature forests to croplands, or increased agricultural products imports (McCarl & Schneider, 2001), emissions than potential sinks from plantations will occur globally.

3.6 Soil Conservation

Soils are the main terrestrial carbon sink. By conserving soil carbon, we can reduce CO_2 emissions and contribute to climate change mitigation. According to The Royal Society (2001), carbon stored in soils is three times the carbon stored as living biomass (1,750 versus 550 PgC). Forests store around 50% of total soil carbon while representing only the 30.3% of emergent lands (FAO, 2006). Soils contain the largest carbon stock in terrestrial ecosystems, representing 50.62% of total carbon in tropical forests, 62.75% in temperate forests and 84.31% in boreal forests (Fujimori, 2001).

The soil carbon pool and associated dynamic processes have not been studied to the same extent, globally, as carbon in living or dead biomass. However, conservation measures, including fire prevention and control, must be developed implemented to conserve carbon pools in soils, because of its high importance to overall carbon storage. Forest harvesting operations commonly result in short-term losses of carbon from the soil (Turner & Lambert, 2000). In fact, studies showed that changes in soil carbon are related to intensity of management (i.e. removal or maintenance of slash, soil compaction or increased radiation due to open canopies), although changes would not be significant over the longer term (Henderson et al., 1995, cited in Paul et al., 2002). Adequate management of harvest debris, which

---

[1] “Revised Approved Afforestation and Reforestation Baseline Methodology Case AR-AM00010 Facilitating Reforestation for Guangxi Watershed Management in Pearl River Basin, China” and “Case ARNM0018, Assisting natural regeneration on degraded land in Albania”, http://cdm.unfccc.int/methodologies/ARmethodologies/approved_ar.html, accessed October 2007.

contain between 20% to 35% of total tree carbon content, is crucial to maintain soil carbon levels.

Although carbon pools in old-growth forests are considered to be in a steady state, Zhou et al. (2006) showed that, between 1973 and 2003, soil organic carbon increased at an average rate of 0.035% each year in old-growth stands (over 400 years old) in China. Based on these results, a longer rotation length may increase soil carbon, even though living biomass accumulation may have reached an asymptote.

Regarding afforested areas, Paul et al. (2002) reviewed global data on changes in soil carbon following afforestation based on 43 previous studies. On average, they found a decrease in soil carbon in the upper soil layer (<30 cm) during the first five years after afforestation, with a recovery to the previous soil carbon levels after 30 years.

4 Conclusions

Forest management practice options available to reduce emissions and/or increase carbon stocks can be grouped in four general strategies (adapted from Nabuurs et al., 2007):

1. Maintaining or increasing the forest area by reducing deforestation and degradation, and through increased areas of plantations or natural expansions of forest land (e.g., afforestation of abandoned lands)
2. Maintaining or increasing the stand-level carbon density through application of appropriate silviculture techniques (e.g., thinning, partial harvests, species compositions, etc.)
3. Maintaining or increasing the landscape-level carbon density through forest conservation, longer rotations, fire management, and pest and disease control
4. Increasing off-site carbon stocks in wood products and enhancing fuel and products substitution by forest-based products (e.g., biomass, building materials, etc.)

Future changes in climate could impact forest growth responses dramatically and modify all the scenarios currently analyzed. In the face of uncertainty, adaptive management could help to develop adequate, operational forestry strategies in a world of constant social and ecological changes (Nyberg, 1998). An increase in the frequency of droughts, or conversely, floods, or a different inter-annual distribution of rainfall could have particular impacts. Although a generally positive effect on forest growth has been stated for different future climate scenarios (Sabaté et al., 2002), local drought and changes to temporal and spatial rainfall distributions may make timber production and carbon storage difficult.

Combined effects of reduced deforestation and degradation, afforestation, forest management, agro-forestry and bio-energy have the potential to increase in the near future (IPCC, 2007) contributing to climate change mitigation and sustainable development.

References

Balboa-Murias, M.A., Rodríguez-Soalleiro, R., Merino, A., Álvarez-González, J.G. (2006). Temporal variations and distribution of carbon stocks in aboveground biomasa of radiata pine and maritime pine pure stands Ander different silvicultural alternatives. *Forest Ecology and Management 234*:29–38.

Blanco, J.A., Imbert, J.B., Castillo, F.J. (2006). Influence of site characteristics and thinning intensity on litterfall production in two *Pinus sylvestris* L. forests in the western Pyrenees. *Forest Ecology and Management 237*(1–3):342–352.

Bogino, S., Bravo, F., Herrero, C. (2006) Carbon dioxide accumulation by pure and mixed woodlands of *Pinus sylvestris* L. and *Quercus pyrenaica* Willd. in Central Mountain Range Spain. In: Bravo, F. (ed.), Proceedings of the IUFRO meeting on managing forest ecosystems: the challenges of climate change, Palencia.

Bravo, F., Díaz-Balteiro, L. (2004) Evaluation of new silvicultural alternatives for Scots pine stands in northern Spain. *Annals of Forest Science 61*(2):163–169.

Bravo, F., Bravo-Oviedo, A., Díaz-Balteiro, L. (2008) Carbon sequestration in Spanish Mediterranean forests under two management alternatives: a modeling approach. *European Journal of Forest Research 127*(3):225–234 DOI 10.1007/s10342–007–0198-y

Bravo, F., Herrero, C., Ordóñez, C., Segur, M., Gómez, C., Menéndez, M., Cámara, A. (2007). Cuantificación de la fijación de CO_2 en la biomasa arbórea de los sistemas forestales españoles. In Bravo, F. (coord.) El papel de los bosques españoles en la mitigación del cambio climático, *pp*. 143–196.

Chiti, T., Publisi, A., Certini, G., Sanesi, G., Buresti, E. (2003). Sequestro del carbonio in suoli su discarich minerarie riforestate con specie azotofissatrici e non. *SISEF atti* 4:53–57.

Ciais, Ph., Reichstein, M., Viovy, N., Granier, A., Ogee, J., Allard, V., Aubinet, M., Buchmann, N., Bernhofer, Chr., Carrara, A., Chevallier, F., De Noblet, N., Friend, A.D., Friedlingstein, P., Gruenwald, T., Heinesch, B., Keronen, P., Knohl, A., Krinner, G., Loustau, D., Manca, G., Matteucci, G., Miglietta, F., Ourcival, J.M., Papale, D., Pilegaard, K., Rambal, S., Seufert, G., Soussana, J.F., Sanz, M.J., Schulze, E.D., Vesala, T., Valentini, R. (2005). Europe-wide reduction in primary productivity caused by the heat and drought in 2003 *Nature 437*(7058):529–533.

Fang, J., Chen, A., Peng, C., Zhao, S., Ci, L. (2001). Changes in forest biomass carbon storage in China between 1949 and 1998. *Science 292*:2320–2322.

FAO (2006). Global Forest Resources Assessment 2005. Progress towards sustainable forest management. FAO Forestry Paper 147. FAO, Rome, 320 pp (http://www.fao.org/docrep/008/a0400e/a0400e00.htm, last accessed September 30 2007).

Franklin, J., Berg, D., Thomburgh, D., Tappeiner, J. (1997) Alternative silvicultural approaches to timber harvesting: variable retention. In: Kohm, D., Franklin, J. (eds.), Creating a forestry for the 21st century. Island Press, Washington, DC, pp. 111–139.

Fuhrer, J., Benitson, M., Fischlin, A., Frei, Ch., Goyette, S., Jasper, K., Pfister, Ch. (2006) Climate risks and their impact on agriculture and forestry in Switxerland *Climatic Change 79*:79–102.

Fujimori, T. (2001). Ecological and Silvicultural strategies for sustainable forest management. Elservier, Amsterdam, The Netherlands, 398 pp.

Gracia, C., Sabaté, S., López, B., Sánchez, A. (2001). Presente y futuro del bosque mediterráneo: balance de carbono, gestión forestal y cambio global. En: Zamora, R., Puignaire, F.I. (eds.) Aspectos funcionales de los ecosistemas mediterráneos. CSIC-AEET, Granada, pp. 351–372.

Gracia, C., Gil, L., Montero, G. (2005). Impactos sobre el sector forestal. En: Moreno Rodríguez, J.M. (ed.) Evaluación preliminar de los impactos en España por efecto del cambio climático. Ministerio de Medio Ambiente, pp. 399–436.

Helms, J.A. (ed.) (1998). The dictionary of forestry. Society of American Forestry, Bethesdam Naryland, 210 pp.

Houghton, R.A., Hackler, J.L., Lawrence, K.T. (1999). The U.S. Carbon Budget: contributions from Land-use change. *Science 285*:574–578.

Ibáñez, J.J., Vayreda, J., Gracia, C. (2002). Metodología complementaria al Inventario Forestal Nacional en Catalunya.. En: Bravo, F., del Río, M., del Peso, C. (eds.), El Inventario Forestal Nacional. Elemento clave para la Gestión Forestal Sostenible, pp. 67–77.

IPCC (2001). Tercer Informe de Evaluación. Cambio climático 2001. Mitigación. Resumen para responsables de políticas y resumen técnico.

IPCC (2007). Summary for policymakers. In: Climate change 2007: mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (http://www.ipcc.ch/SPM040507.pdf, last accessed September 30 2007).

Kaipainen, T., Liski, J., Pussinen, A., Karjalainen, T. (2004). Managing carbon sinks by changing rotation length in European forests. *Environmental Science & Policy 7*:205–219.

Kurz, W.A., Apps, M.J. (1999) A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. *Ecological Applications 9*:526–547.

Kurz, W.A., Beukema, S.J., Apps, M.J. (1998) Carbon budget implications of the transition from natural to managed disturbance regimes in forest landscapes. *Mititgation and Adaptation Strategies for Global Change 2*:405–421.

Lindner, M., Karjalainen, T. (2007). Carbon inventory methods and carbon mitigation potentials of forests in Europe: a short review of recent progress. *European Journal of Forest Research 126*:149–156.

Liski, J., Perruchoud, D., Karjalainen, T. (2002). Increasing carbon stocks in forest soils of western Europe. *Forest Ecology and Management 169*:159–175.

Liu, J., Peng, C., Apps, M., Dang, Q., Banfield, E., Kurz, W. (2002). Historic carbon budgets of Ontario's forest ecosystems. *Forest Ecology and Management 169*:103–114.

Martin, E. (2005). Revisión de la ordenación del M.U.P. n° 50 "Pinar del Llano" del término municipal de Portillo (Valladolid). Proyecto Fin de Carrera, Ingeniería Técnica Forestal. E,T.S. de Ingenierías Agrarias de Palencia. University of Valladolid, Spain.

McCarl, B.A., Schneider, U.A. (2001) Greenhouse gas mitigation in U.S. agriculture and forestry. *Science 294*:2481–2482.

Melillo, J., Prentice, I.C., Farquhar, C.D., Schulze, E.D., Sala, O.E. (1996). Terrestrial biotic responses to environmental change and feedbacks to climate. In: Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K. (eds.), Climate change 1995-the science of climate change. Cambridge University Press, Cambridge, pp. 445–481.

Montero, G., Muñoz, M., Donés, J., Rojo, A. (2004). Fijación de CO_2 por Pinus sylvestris L. y Quercus pyrenaica Willd. En los montes "Pinar de Valsaín" y "Matas de Valsaín". *Investigación Agraria: Sistemas y Recursos Forestales 13*(2) 399–416.

Montero, G., Ruiz-Peinado, R., Muñoz, M. (2005). Producción de biomasa y fijación de CO_2 por los bosques españoles. Monografías INIA: Serie Forestal, Madrid, 270 pp.

Nabuurs, G.-J., Schelhass, M.-J., Mohren, M.J., Field, C.B. (2003). Temporal evolution of the European forest sector carbon sink from 1950 to 1999. *Global Change Biology 9*:152–160.

Nabuurs, G.-J., Masera, O., Andrasko, K., Benítez-Ponce, P., Boer, R., Dutsckhe, M., Elsiddig, E., Ford-Robertson, J., Frumhoff, P., Karjalainen, T., Krankina, O., Kurz, W.A., Natsumoto, M., Oyhantcabal, W., Ravindranath, N.H., Sanz Sánchez, M.J., Zhang, X. (2007). Forestry. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (eds.), Climate change 2007: mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Nyberg, J.B. (1998). Statistics and the practice of adaptive management. In: Sit, V., Taylor, B. (eds.), Statistical methods for adaptive management studies. British Columbia Ministry of Forests, Research Branch, Land Management Handbook No. 42, Victoria, British Columbia, Canada, pp. 1–7.

Osorio, L.F., Pando, V., Del Peso, C., Bravo, F. (2006). Long-Term implications of traditional forest regulation methods applied on Maritime pine *Pinus* pinasterAit. forests in Central Spain. Unpublished manuscript.

Pacala, S.W., Hurtt, G.C., Baker, D., Peylin, P., Houghton, R.A., Birdsey, R.A., Heath, L., Sundquist, E.T., Stallard, R.F., Ciais, P., Moorcroft, P., Caspersen, J.P., Shevliakova, E., Moore, B., Kohlmaier, G., Holland, E., Gloor, M., Harmon, M.E., Fan, S.-M., Sarmiento, J.L., Goodale, C.L., Schimel, D., Field, C.B. (2001). Consistent land- and atmosphere-based U.S. carbon sink estimates. *Science 292*:2316–2320.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K. (2002). Change in soil carbon following afforestation. *Forest Ecology and Management 168*:241–257.

Pohjola, J., Valsta, L. (2006). Carbon credits and management of Scots pine and Norway spruce stands in Finland. *Forest Policy and Economics 9*(7):789–798.

Reichstein, M., Tenhunen, J.D., Roupsard, O., Ourcival, J.M., Rambal, S., Miglietta, F., Peressotti, A., Pecchiari, M., Tirone, G., Valentini, R. (2002) Severe drought effects on ecosystem CO_2 and H_2O fluxes at three Mediterranean evergreen sites: revision of current hypothesis? *Global Change Biology 8*:999–1017.

Río, M. del (1999). Régimen de claras y modelos de producción para Pinus sylvestris L. en los sistemas Central e Ibérico. Tesis doctoral, INIA, 257 p.

Río, M. del, Barbeito, I., Bravo-Oviedo, A., Calama, R., Cañellas, I., Herrero, C., Bravo, F. (2008). Carbon sequestration in Mediterreanean pine forests. In: Bravo, F., LeMay, V., Jandl, R., Gadow, K. von (eds.), Managing forest ecosystems: the challenge of climate change.

Sabaté, S., Gracia, C.A., Sanchez, A. (2002). Likely effects of climate change on growth of Quercus ilez, Pinus halepensis, Pinus pinaster, Pinus sylvetris and Fagus sylvatica forest in the Mediterranean region. *Forest Ecology and Management 162*(1): 23–37.

Scarascia-Mugnozza, G., Matteucci, G., Montagnani L., Masci, A. (2001). Gestione forestale sostenibile e carbonio organico nei suoli in ambiente mediterraneo: Inquadramento del problema e aspetti metodologici per una ricerca nel territorio del Parco Nazionale della Calabria. *L'Italia Forestale e Montana*.

Schmid, S., Thürig, E., Kaufmann, E., Lischke, H., Bugmann, H. (2006). Effect of forest management on futer carbon pools and fluxes: A model comparison. *Forest Ecology and Management 237*:65–82.

Sohngen, S., Andrasko, K., Gytarsky, M., Korovin, G., Laestadius, L., Murray, B., Utkin, A., Zamolodchikov, D. (2005) Stocjs and flows: carbon inventory and mitigation potential of the Russian forest and land base. Report of the World Resources Institute, Washington, DC.

The Royal Society (2001). The role of land carbon links in mitigating global climate change. Policy document 10/01 35 pp.

Turner, J., Lambert, M. (2000). Change in organic carbon in forest plantation soils in eastern Australia. *Forest Ecology and Management 133*:231–247.

Woodbury, P.B., Smith, J.E., Heath, L.S. (2007).Carbon sequestration in the U.S. forest sector from 1990 to 2010. *Forest Ecology and Management 241*:14–27.

Zhou, G., Liu, S., Li, Z., Zhang, D., Tang, X., Zhou, C., Yan, J., Mo, J. (2006). Old-Growth forests can accumulate carbon in soils. *Science* 314:1417.

Analysing the Effects of Forest Structure on Carbon Stocks and Timber Production Under Changing Management and Climate

J. Garcia-Gonzalo[1]*, H. Peltola[2], A. Zubizarreta Gerendiain[3], and S. Kellomäki[4]

1 Introduction

The growth of boreal forests in northern Europe is currently limited by a short growing season, low summer temperatures and short supply of nitrogen (Kellomäki et al., 1997; Nohrstedt, 2001; Olsson, 2006). Forest development is expected to be affected by the expected change in climate in response to the rapid increase of greenhouse gases in the atmosphere, particularly carbon dioxide (CO_2) (IPCC, 2001). In Finland, the future climate is anticipated to be an increase of 2–7°C in annual mean temperature (T) and an increase of 6–37% in precipitation with a concurrent doubling of CO_2 by 2100 (Carter et al. 2002). The increase in T may prolong the growing season and also enhance the decomposition of soil organic matter (SOM), thereby increasing the supply of nitrogen (Melillo et al. 1993; Lloyd & Taylor, 1994). These changes may substantially enhance forest growth, timber yield and the accumulation of carbon (C) in the boreal forests (Giardina & Ryan, 2000; Jarvis & Linder, 2000; Luo et al., 2001; Strömgren, 2001).

Previous model-based studies applying current management regimes ("business-as-usual") showed an increase in growth, yield and C stocks in the boreal forests (both at stand/regional level) under a changing climate (e.g. Pussinen et al., 2002). However, there may be a need to adapt management for the altered dynamics of the forest ecosystem (Lindner, 2000; Fürstenau et al., 2007, Garcia-Gonzalo et al., 2007a, b), in order to avoid harmful effects on forests and to optimally utilize the anticipated increased growth and yield under these climate changes. As well as changes in management, the structure of the forest landscape, particularly the

---

[1] Faculty of Forest Sciences, University of Joensuu (also partly at the European Forest Institute, EFI), tel.: +358132513639, *jordi.garcia@joensuu.fi

[2] Senior lecturer for Silvicultural Sciences at the University of Joensuu, Faculty of Forest Sciences

[3] Ph.D. candidate at the University of Joensuu, Faculty of Forest Sciences

[4] Dean and Professor for Silvicultural Sciences at the Faculty of Forest Sciences, University of Joensuu

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

age-class distribution, is one of the key factors affecting the timber yield and C stocks in managed forests (Garcia-Gonzalo et al., 2007c). Newly regenerated sites probably have a net emission of C, whereas young stands sequester C. In maturing stands, the C sequestration is reduced along with declining growth, and over-mature stands may even result in a net C emission (Jarvis et al., 2005). Therefore, for sustainable management, a forest landscape should be comprised of stands representing different development stages to maintain timber production and C sequestration in the forest ecosystem. Also, differences in timber productivity and C stocks are expected for differences in species compositions over the landscape (e.g. Briceño-Elizondo et al., 2006a, b). Overall, since climatic conditions influence the growth and development of trees and stands, climate changes are expected to affect the dynamics, and therefore timber production and C sequestration on forest landscapes (Garcia-Gonzalo et al., 2007c).

Empirical growth and yield models are widely used to support forestry decision-making. Usually, these models utilize historical data such as forest inventory data and are not capable of modeling growth responses to changes in growing conditions. Gap or Patch models describe the reproduction, growth and mortality of trees (e.g. Botkin, 1993), and interactions among species. These models commonly explicitly assess the impact of temperature, water and nutrients on tree growth, but the main objective is to model changes in species composition (succession) over time. These models have been used to assess the potential vegetation patterns and changes in vegetation distribution under expected climate change. Mechanistic or process-based models are based on physiological processes controlled by climatic and edaphic factors, with the main goal of describing the interaction between plants and the surrounding environment (Waring & Running, 1998). Consequently, this type of model is designed to be responsive to changes in environmental conditions. The use of process-based models in forestry decision-making has been limited since these models may require input data not provided by conventional forest inventories. However, process-based model can provide the similar predictions under practical management situations as empirical models (Matala et al., 2003). Moreover, process-based models may help to understand how forests grow and develop under a changing environment (Sands et al., 2000), and help to select management regimes that may avoid detrimental impacts and utilize the opportunities probably provided by the climate change (Lindner, 2000).

In selecting amongst alternative management regimes, one of the main concerns is that a preference for enhancing C stocks might induce opportunity costs for timber production. Thus, it is important to investigate how to manage forests in a sustainable way to balance timber production and C sequestration. In particular, a study of how forest structure, in terms of age class distribution and species composition over the landscape, affects timber production and carbon stocks is needed.

In this study, we used the process-based model (FINNFOR) to assess the sensitivity of forest productivity to the structure (area of each species and age class distribution over the unit) of a boreal forest landscape under changing management and climatic conditions. Forest productivity was studied over 100 years in terms of timber production (harvested timber, total net present value NPV) and C stocks (C in the ecosystem, in trees and in soil). Moreover, the opportunity cost of C sequestration by C sink enhancement was calculated as the potential loss in the timber NPV when management for maximum C stocks was selected over management for maximum timber NPV. The study utilized actual ground measurements from a selected forest management unit representing typical forests growing in central and eastern Finland as a basis for the computations.

2 Material and Methods

2.1 Model Used in Simulations

The process-based model (FINNFOR) originally developed by Kellomäki and Väisänen (1997) was selected for use in this study. The model provides predictions of photosynthetic production, growth, timber yield and carbon balance (C in trees and in soil) of stands in response to different climate conditions and different management regime used in comparing responses of different management regimes under current and changing climates (see Strandman et al., 1993; Kellomäki et al., 1997; Kellomäki & Väisänen, 1997; Kramer et al., 2002; Matala et al., 2003).

The physiological core of the model is the biochemical sub-model of photosynthesis developed by Farquhar et al. (1980) and von Caemmerer and Farquhar (1981). The dynamics of the forest ecosystem are directly linked to climate through photosynthesis, respiration and transpiration. Furthermore, hydrological and nutrient cycles indirectly couple the dynamics of the ecosystem to climate change through soil processes, which represent the thermal and hydraulic conditions in soil, and the decomposition of litter and humus with the mineralization of nitrogen (Chertov & Komarov, 1997). The gross photosynthesis over a year provides the total amount of photosynthates available for the maintenance and growth of object trees and tree components (foliage, branches, stem, coarse and, fine roots). The allocation of the available photosynthates among the component is based on the allometry between the mass of each component and the total mass of the object tree (Matala et al., 2003). The mass of the stem and its growth is used to annually calculate diameter at breast height of the stem (DBH) (1.3 m above ground level) and height of object trees. These calculations are performed for each cohort, defined by the tree species, the number of trees per hectare, diameter (cm), height (m) and age (year). These variables are used as the inputs of the initial stand data for the simulations and are updated annually during the simulation.

Stocking, in terms of the level of occupation by trees, controls the dynamics of the ecosystem through changes in mortality and growth rates. Management also modifies the forest structure, which changes the canopy processes and availability of resources for physiological processes, thereby affecting forest growth. Management includes regeneration through planting, thinning of trees in stands, and selection of the rotation length. In FINNFOR, thinning is simulated via basal area reduction, converted into the number of trees to be removed from each tree cohort. Trees removed in thinning and in final harvests are converted to saw logs and pulp wood using empirical tables as a function of DBH and height (Laasasenaho & Snellman, 1983). These tables provide the amount of saw logs, pulp wood and logging residue as a function of the DBH and tree height.

Model parameterization of FINNFOR and the performance of its predictions have been reported earlier in detail in several papers: (i) model parameterization based on data from long-term forest ecosystem and climate change experiments by Kellomäki et al. (2000), (ii) model validation against growth and yield tables by Kellomäki and Väisänen (1997), (iii) measurements of short-term stand-level fluxes of water and carbon by means of the eddy covariance method, along with model evaluation against five other process-based models by Kramer et al. (2002) and (iv) measurements of the growth history of trees in thinning experiments by Matala et al. (2003). Matala et al. demonstrated that FINNFOR is capable of reconstructing the growth history of Scots pine, Norway spruce and birch. In previous studies, parallel simulations and model comparisons were performed using FINNFOR and the empirical growth and yield model called Motti, developed by Hynynen et al. (2002) using tree growth data measured on a large number of forest inventory sample plots in Finland (see Matala et al., 2003; Briceño-Elizondo et al., 2006a). Also, sensitivity analyses of FINNFOR outputs (e.g. growth, timber yield and C stocks) to different climate parameters and to the initial structure of the forest have been carried out (Briceño-Elizondo et al., 2006a; Garcia-Gonzalo et al., 2007a, b, c). In summary, these previous studies indicated that the FINNFOR model performs in an acceptable way and has similar capacities for prediction as an accepted empirical model.

2.2 *Model Simulations*

2.2.1 Layout for the Simulations

The management unit used in this study as a basis for model-based analyses is located in eastern Finland (63°01' N 27°48' E, the mean altitude is 94 m above sea level). The study area included 1,018 separate stands covering 1,451 ha (based on 2001 inventory). The breakdown of stands was: Norway spruce (*Picea abies*) dominated stands, 64% of the total area (933 ha); Scots pine (*Pinus sylvestris*) dominated stands, 28% (412 ha); and silver birch (*Betula pendula*) dominated stands, 7% (106 ha). The sites were of *Oxalis Myrtillus* (OMT), *Myrtillus* (MT) and *Vaccinium* (VT) types (Cajander, 1949). The dominant tree species on the fertile sites (OMT, MT) was Norway spruce, while the main tree species on the poor sites (VT) was mostly Scots pine. Each stand was described in terms of the dominant tree species, average stand age, mean height and mean DBH, stand density and site type.

Since simulation time for 1,018 stands under different management regimes and forest structures would be very long, representative stands were selected. The stands were classified into groups representing the same dominant tree species, age class (10 years intervals; 0–100 years) and site type. Then, a typical stand was selected from each group. In total, 42 representative stands were selected for the simulations. In each representative stand, the number of trees was divided equally into three cohorts. In the first cohort, the average DBH and tree height were those obtained in the inventory. In the second cohort, the average DBH and height were increased by 15%, while in the third cohort the values were reduced by 15%. The initial mass of organic matter in the soil was assumed to be 70 Mg ha^{-1}.

For analyses of the effects of age class distributions on timber production and C stocks over the study area, four different age class distributions (distributions A, B, C, D) were created by changing the original distribution over the management unit. For this purpose the following age classes were used: (1) sapling stands, 0–20 years old; (2) stands ready for first thinning, 21–40 years old; (3) stands ready for final thinning, 41–70 years old; and (4) mature stands >70 years old, ready for a final regeneration harvest. Consequently, the following distributions were applied by modifying the area originally occupied bye each of the age classes (Table 1): A: distribution dominated by intermediate age classes (normal distribution); B: uniform distribution by age class; C: distribution dominated by young age classes (left-skewed distribution); and D: distribution dominated by old age classes (right-skewed distribution).

Similarly, the effects of the proportions of different species over the landscape on timber production and C stocks were simulated. Three different scenarios were simulated: (1) Case I, 60% Norway spruce, 30% Scots pine, 10% silver birch, resembling the original species distribution in the management unit; (2) Case II, 30% Norway spruce, 60% Scots pine, 10% silver birch, interchanging the proportion of area covered by Norway spruce and Scots pine; and Case III, 45% Norway spruce, 45% Scots pine, 10% silver birch, an equal area covered by the two conifers. The proportion of silver birch was not changed since this simulated the preferences of the forest owners in planting conifers. Cases II and III were created to indicate the results of increasing the area occupied by Scots pine in regard to Norway spruce. Two other cases were simulated, where the total area was covered by each of the two conifers, but results are not presented here.

Table 1 Percentage of area occupied by each of the age class groups and species. Descriptions of the cases are given in the text

		Age class distribution			
Age class groups	Age (years)	A (Normal) (%)	B (Uniform) (%)	C (Left) (%)	D (Right) (%)
Sapling stands	0–20	25	25	50	10
Stands ready for first thinning	21–40	30	25	25	15
Stands ready for second thinning	41–70	30	25	15	25
Stands ready for final cutting	>70	15	25	10	50

2.2.2 Management Scenarios

In constructing the thinning regimes for the analysis, the current management recommendations (Yrjölä, 2002) were used to define the "business-as-usual" management (Basic thinning, BT(0,0)) which later was altered for the remainder of the simulated management regimes. The recommendations guidelines are species-specific, and they use the dominant height and basal area to defining the timing and intensity of thinning (Fig. 1). When the upper limit for basal area (thinning threshold) at a given dominant height is encountered, thinning is triggered. Thus, the timing of thinning was adjusted for the growth and development of the stand and took place before mortality due to crowding. The timings of thinnings were therefore not at pre-defined ages. The minimum dominant height for thinning was set to 12 m, since trees are not susceptible to natural mortality due to crowding prior to this height. Random mortality also occurred before and after the threshold value of the dominant height. Because most of the sites were of the MT and OMT types (83%), thinning rules recommended for these two types were used for all site types stand. All thinnings were from below by removing mainly smaller suppressed trees at each intervention.

To simulate different management scenarios, BT(0,0) can be varied by altering the thinning threshold and the residual basal area following thinning. Based on a previous studies at the stand level by Briceño-Elizondo et al. (2006b) and at unit level by Garcia-Gonzalo et al. (2007a), two other thinning regimes were simulated concurrently increasing the thinning threshold and the remaining basal area after thinning by either 15% or 30% ((BT(15,15) and BT(30,30)) over BT(0,0). An unthinned (UT(0,0)) regime was also simulated, for a total of four management regimes.

The simulations for the representative stands covered a 100-year period. Regardless of tree species and site type and for all management regimes, the stand was clear-cut (i.e. regeneration harvest or final cut) at 100 years, or when the average DBH exceeded 30 cm, following Yrjölä (2002). If a stand was clear-cut before the end of the 100-year simulation period, the stand was planted with the same species that occupied the site prior to the final cut, at a planting density of 2,500 saplings

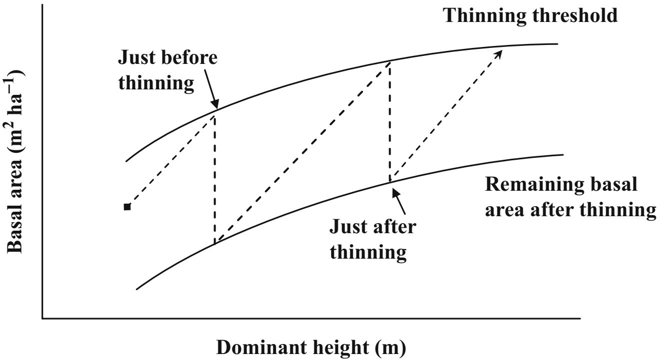


Fig. 1 Application of dominant height and basal area in governing the timing of thinning

per hectare for all species and site types. Once the stand was established, the simulation continued to the end of the 100-year period.

2.2.3 Climate Scenarios

The two climate change scenarios simulated were the current climate and one climate change scenario compiled by the Potsdam Institute for Climate Impact Research (PIK), Germany. The weather data for the period 1961–1990 represented the "current" climate over the period 2000–2100 by repeating the data for this 30-year period from year to year over the whole simulation period. The climate change scenario used was the HadCM2 scenario following the model prediction derived from the Hadley Centre Global Circulation Model (GCM) (Erhard et al., 2001; Sabaté et al., 2002). The daily weather statistics from the different climate scenarios were scaled down to an hourly basis using the weather simulator developed by Strandman et al. (1993). In the scenario representing the current climate, the annual mean temperature and precipitation in the last 30 years (2071–2100) were 3.1°C and 478 mm year^{-1}, whereas for the HadCM2 scenario, the corresponding values were 7.2°C and 560 mm year^{-1} (Fig. 2). Under the current climate, the CO$_2$ concentration used was 350 ppm. For the HadCM2 scenario, an increase up to 653 ppm over the period 2000–2100 was used. The increase was smaller during the early phases of simulations than in the latter phases.

2.2.4 Simulation Outputs and Data Analyses

The C stocks in trees and soil were calculated in terms of the mean C storage over the simulation period (Mg C ha^{-1}), and the growth of stem wood and timber

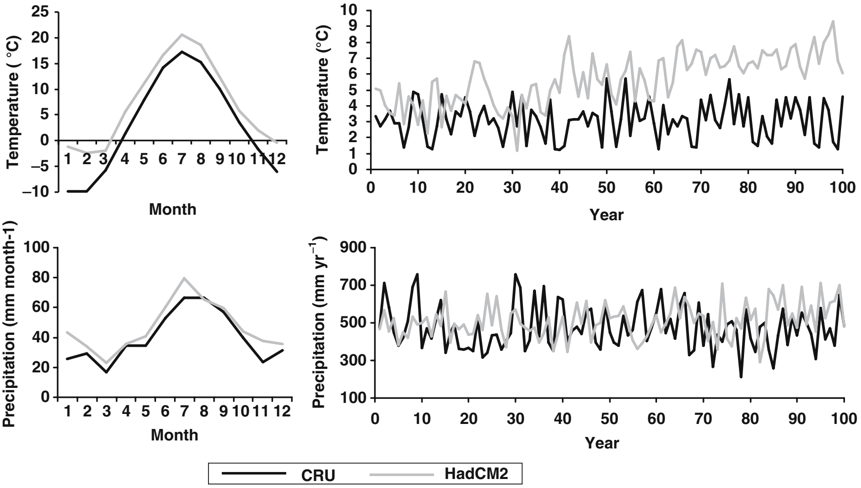


Fig. 2 Variation in mean annual and mean monthly air temperature and precipitation over the simulation period for the current climate (CRU) and changing climate (HadCM2) scenarios under study

yield (saw logs and pulp wood from thinnings and final harvest) were considered to be the total for the 100 year simulation period (m^3 ha^{-1}). All the variables were calculated over the management unit applying different forest structures, management regimes and climate scenarios. Furthermore, the incomes and costs (e.g. planting and other regeneration costs) were included in the analysis by calculating the net present value (NPV) which was calculated as the discounted value of the future expected net cash flow from the forest timber (Cf) plus the discounted liquidation value of the standing stock at the end of the simulation time (LV_{100}) for the management unit applying the discount rates (p) of 1, 3 and 5% (see Eq. 1).

$$NPV = \sum_{t=1}^{100} \frac{Cf_t}{(1+p)^t} + \frac{LV_{100}}{(1+p)^{100}} \tag{1}$$

In these calculations, prices of different timber assortments per species were the average stumpage prices (€ m^{-3}) for the period 1990–2000 (Finnish Statistical Yearbook of Forestry, 2001). For silver birch, Scots pine and Norway spruce, the prices of saw logs were 41€ m^{-3}, 41€ m^{-3} and 34€ m^{-3}, respectively, and for pulp wood 14€ m^{-3}, 15€ m^{-3} and 19€ m^{-3}. Similarly, costs of the regeneration operations (soil preparation and plantation per species) were set to the average prices for the period 1990–2000 for Finnish conditions: (i) harrowing and scarification techniques were assumed to be used in soil preparation for Scots pine (140€ ha^{-1}), while ploughing and mounding were assumed for Silver birch and Norway spruce (226€ ha^{-1}); and (ii) planting costs were assumed to be 712€ ha^{-1} regardless of species. The different ecosystem compartments used in the analysis (C stocks in trees and soil) were calculated in terms of the mean C stock over the 100 year simulation (Mg C ha^{-1}).

Based on the NPV and mean C stocks in the ecosystem during the 100-year period, the potential economic loss that may occur in choosing a management regime that increases C stock was indicated by an opportunity cost of lost timber revenues. In order to estimate the opportunity cost, we assumed that the differences in the NPV among the management and climate scenarios were entirely due to an interaction between the climate and management (i.e. prices and costs are held constant in all the scenarios). In these calculations, carbon fixed in harvested timber and resulting wood products were not included in the accounting. The price of carbon was based on an indirect pricing method based on the opportunity cost that may occur as a result of the reduction in timber production when using management alternatives that increase the C stock. For the calculations, the differences in NPV produced by the change in management were divided by the differences in C stock. Thus, the potential marginal cost of increasing C stock (**potMC**) refers to the differences in the C stock and in NPV of timber representing the management regimes maximizing the C stock and NPV respectively. On the other hand, the current marginal cost (**curMC**) refers the differences in the C stock and in NPV of timber production when one shifts from the current management to that which maximizes the C stock in the forest ecosystem.

3 Results

3.1 Forest Structure Effects on Timber Yield and Net Present Value Under Business-As-Usual Management

Forest structure changing over time as a result of thinning and final cuts, and also because of differences in growth rates related to age class distribution and species differences. The age class distribution changes over time even when starting with a uniform age class distribution. Under current climate, the forest structure with an age distribution initially skewed to the left (i.e. dominated by young stands) became skewed to the right (i.e. dominated by old stands) over 60 years (Table 2). This was the case also in the forest landscape representing the normal and uniform age class distributions, but the change was not as pronounced as in the previous case. At the same period, the initial age class distribution skewed to the right became dominated by stands mature for thinning, with the distribution resembling the normal one. At the end of the 100-year simulation period, the age class distributions became skewed to the left regardless of the initial age class distribution. Thus, the structure of the forest landscape changed dynamically over time, with patterns affected also by varying growth rates of different tree species and management.

Under changing climate, the growth was more rapid and repeated thinning events and final cuts occurred earlier. Consequently, the structural development of the forest landscape after the projection periods differed from that under current climate (Table 2). This was evident even after 60 years of simulation, but the differences between the age class distributions under the current climate and the climate change were especially pronounced at the end of the 100-year simulation period. For all initial age class and species distributions, the age class distributions

Table 2 Percentage of area occupied by each of the age class regardless of the species distribution used for different time steps and depending on the initial age class distribution used (A, B, C or D) when the business-as-usual management regime is used (BT(0,0))

Initial class age distribution	Initial: Year 2001				60 years: Year 2060				100 years: Year 2100			
	Age				Age				Age			
	1–20	21–40	41–70	>70	1–20	21–40	41–70	>70	1–20	21–40	41–70	>70
Current climate (CRU)												
A (normal)	25	30	30	15	18	18	27	36	31	29	28	12
B (uniform)	25	25	25	25	15	17	35	32	37	25	23	14
C (left)	50	25	15	10	11	10	34	45	47	31	16	7
D (right)	10	15	25	50	13	21	48	17	42	14	21	23
Climate change (HadCM2)												
A (normal)	25	30	30	15	28	21	27	23	21	33	41	5
B (uniform)	25	25	25	25	24	20	35	21	21	36	34	9
C (left)	50	25	15	10	20	12	34	34	24	45	27	4
D (right)	10	15	25	50	18	23	48	10	22	33	28	18

under the climate change resembled the normal distribution, whereas the age class distributions were skewed to the left under the current climate. This change was especially pronounced for the simulations where the initial distribution was skewed to the right (dominated by old stands), with the number of stands ready for first thinning being more than double the number under the current climate.

The timber yield predictions over 20-year simulation periods under BT(0,0) were affected by the different age class distributions used (Table 3). Under the current climate and regardless of the species distribution over the unit, the normal and the uniform age class distributions gave a more balanced timber harvest over the time as shown by the coefficient of variation (COV) over the 20-year projection periods (Table 3). As one may expect, the initial age class distribution skewed to the left, dominated by young stands, provided most of the timber during the latter years of the simulation period, whereas the distribution to the right yielded most of the timber early in the simulation period, with both distributions having a very uneven flow of harvests (high COV). Consequently, the initial age class distribution skewed to the right gave the highest NPV for timber produced over the rotation. Any increase in the number of Scots pine dominated stands relative to Norway spruce dominated stands (see Cases II and III) led to a decrease in timber production since Norway spruce is the most productive species in the unit.

Under changing climate, the effect of the forest structure (age class distributions and species distribution) followed the same pattern as under current climate, but timber yields was generally larger and final harvests occurred earlier in the projection period. Consequently the NPV was higher under changing climate than under current climate (Table 3).

Table 3 Total amount of harvested timber yield (m^3 ha^{-1}) and coefficient of variation (COV) of the harvested timber yield (%) over time periods of 20 years for the business-as-usual thinning regime for different age class distributions (A, B, C or D) and proportions of species (three different cases) over the management unit. The four age distributions are: A (normal distribution); B (uniform); C (left-skewed) and D (right-skewed). Case I resembles the current share of species in the unit (60% Norway spruce, 30% Scots pine, 10% silver birch), Case II (30% Norway spruce, 60% Scots pine, 10% silver birch), and Case III (45% Norway spruce, 45% Scots pine, 10% silver birch)

	Case I				Case II				Case III			
Year	A	B	C	D	A	B	C	D	A	B	C	D
Current climate (CRU)												
2020	106	128	74	197	92	113	70	170	99	120	72	184
2040	121	111	114	95	108	102	96	99	115	106	105	97
2060	119	119	102	128	99	96	83	102	109	108	93	115
2080	120	113	120	97	122	113	116	95	121	113	118	96
2100	132	142	153	152	114	119	150	109	123	131	151	130
COV	7.9	10.5	25.6	31.6	11.1	8.5	30.5	27.4	8.7	8.7	27.7	28.9
Climate change (HadCM2)												
2020	111	133	77	202	99	119	74	176	105	126	76	189
2040	137	125	132	105	120	113	108	107	129	119	120	106
2060	139	141	119	157	116	118	98	133	127	130	109	145
2080	160	163	171	164	149	144	148	131	155	154	160	148
2100	137	129	132	116	133	130	162	105	135	130	147	110
COV	12.7	10.9	26.7	26.3	15.2	10.0	30.8	22.0	13.6	10.0	27.0	24.1

3.2 *Effects of Forest Structure and Management on Average Timber Yield and Net Present Value over the 100-Year Simulation Period*

As expected, the average timber yield per hectare over the 100-year simulation period was affected by the initial age class distribution, the share of species in the unit and the management used. Under current climate, regardless of the thinning regime used and the proportions of area by species, the most timber was obtained when the initial forest landscape was dominated by old stands (right-skewed distribution), followed by the landscape with the stands uniformly distributed among the age classes and by the landscape with a normal distribution (Table 4). The distribution skewed to the left (forest dominated by young stands) gave the smallest amount of timber yield. However, this pattern was not valid for the unthinned regime (UT(0,0)). In that case, the age distribution skewed to the left gave the highest amount of timber yield.

The relative proportional areas of Norway spruce and Scots pine dominated the management unit response. For example for 100% Norway spruce under BT(0,0) and a normal age class distribution, the average timber yield over the unit would have been 694 m^3 ha^{-1} compared to 485 m^3 ha^{-1} and 354 m^3 ha^{-1} for 100% dominance by Scots pine and silver birch, respectively. Thus, regardless of management, climate and initial age class distribution, the forest structure mainly dominated by Norway spruce (Case I) produced the highest amount of timber yield. Moreover, the total timber production was reduced when Scots pine area was increased and the Norway spruce was reduced (see Cases I-III in Table 4). On the other hand, regardless of the age class distribution and the proportional area by species in the

Table 4 Total amount of harvested timber (m^3 ha^{-1}) depending on the age class distributions (A, B, C or D) and changing the share of species (cases I, II and III) over the management unit under different management regimes and current and climate change scenarios. The four distributions are: A (normal distribution); B (uniform); C (left-skewed) and D (right-skewed). Case I resembles the current species distribution in the management unit (60% Norway spruce, 30% Scots pine, 10% Silver birch), Case II (30% Norway spruce, 60% Scots pine), Case III (45% Norway spruce, 45% Scots pine, 10% Silver birch)

Management scenarios	Case I				Case II				Case III			
	A	B	C	D	A	B	C	D	A	B	C	D
Current climate (CRU)												
UT(0,0)	484	469	491	432	419	410	420	390	451	440	456	411
BT (0,0)	598	613	562	669	535	543	514	574	566	578	538	622
BT(15,15)	619	638	592	696	560	569	546	598	590	603	569	647
BT(30,30)	650	669	628	725	589	597	580	620	620	633	604	673
Climate change (HadCM2)												
UT(0,0)	546	535	567	499	467	460	481	433	506	497	524	466
BT (0,0)	684	692	632	742	618	623	590	651	651	657	611	697
BT(15,15)	724	731	674	778	653	656	630	675	689	693	652	727
BT(30,30)	762	769	716	814	688	690	671	703	725	729	693	759

unit, the timber yield tended to increase if thinning was delayed and more basal area was retained (i.e., BT(15,15) and BT(30,30) higher than BT(0,0)). Under current climate, the maximum yield was obtained under BT(30,30).

Under the changing climate, the effects of age class distribution, proportional area by species and management regime remained similar to under current climate conditions, but the timber yield increased up to 11–17% depending on the management regime and the forest structure (Table 4). Regardless of the species distribution, for the thinned stands the maximum increase was found for the normal distribution and the minimum for the right-skewed distribution. For unthinned stands, the highest increase of timber yield due to changing climate was for the left skewed age distribution, and the smallest was for the normal distribution.

Under current climate, taking BT(0,0) as the baseline management scenario, Case I of species distribution (60% Norway spruce, 30% Scots pine, 10% silver birch) and a discount rate of 3%, the highest NPV (7,206€ ha^{-1}) was found when the forest structure was mainly dominated by old stands ready for regeneration (right-skewed distribution) (Table 5). NPV for a uniform age distribution was 5,507€ ha^{-1}, for a normal distribution was 5,068€ ha^{-1} and for a left-skewed distribution was 3,887€ ha^{-1}. This effect of age class distribution was shown for all the different management regimes, and regardless of the discount rates and the species distributions compared. Furthermore, regardless of management and initial age class distribution used the species proportions in the unit affected the total NPV. If the management unit had been fully occupied by Norway spruce the NPV (with $p = 3\%$) would have been 5,775€ ha^{-1} under BT(0,0) with a normal age class distribution compared to NPV of 4,388€ ha^{-1} for 100% dominance of Scots pine. Thus, the share of species over the management unit affected the results substantially, and the NPV was reduced when Scots pine area was increased and the Norway spruce was reduced (i.e., Cases I-III).

Any increase in timber yield due to a change in management corresponded with a similar increase in the NPV (Table 5). Regardless of the initial age class distribution, the unthinned management regime gave the smallest profit. The NPV tended to increase if thinning was delayed and the retained basal area was higher (i.e., BT(15,15) and BT(30,30) higher than BT(0,0). Sensitivity analysis of the NPV

Table 5 Net present value (€ ha^{-1}) for a discount rate of 3%, under current climate for different management regimes depending on the age class distribution and changing the proportion of species over the unit. The four age class distributions are: A (normal distribution); B (uniform); C (left-skewed), D (right-skewed). Case I resembles the current share of species in the unit (60% Norway spruce, 30% Scots pine, 10% silver birch), Case II (30% Norway spruce, 60% Scots pine, 10% silver birch), Case III (45% Norway spruce and Scots pine, 10% silver birch)

Management	Case I				Case II				Case III			
scenarios	A	B	C	D	A	B	C	D	A	B	C	D
UT(0,0)	3,884	4,401	2,807	6,248	3,431	3,925	2,485	5,632	3,657	4,163	2,646	5,940
BT (0,0)	5,068	5,507	3,887	7,206	4,652	5,053	3,627	6,577	4,860	5,280	3,757	6,891
BT(15,15)	5,191	5,619	4,093	7,273	4,749	5,135	3,780	6,604	4,970	5,377	3,936	6,939
BT(30,30)	5,214	5,646	4,136	7,292	4,785	5,174	3,836	6,632	5,000	5,410	3,986	6,962

calculation revealed that the discounting rate (p) had a strong influence on the NPV under all the management regimes regardless of the proportions of species (Fig. 3). In the case of discount rates of 5%, the NPV for BT(0,0) was similar or even higher than for other management regimes (Fig. 3).

Focusing solely on the climate change effects, the NPV increased regardless of the management regimes, age class and species distributions used (Fig. 4). The rate of increase of NPV due to climate change was similar for any of the species distributions compared (Fig. 4). However, the level of climate change impact was dependent on the discount rate. The effect of management under climate change was in the line of the findings under current climate conditions. Regardless of the species distribution and management regime used, high differences in the predictions of climate change effect were found depending on the initial age class distribution used. The maximum increase was found for the left-skewed distribution and the minimum for the right-skewed distribution.

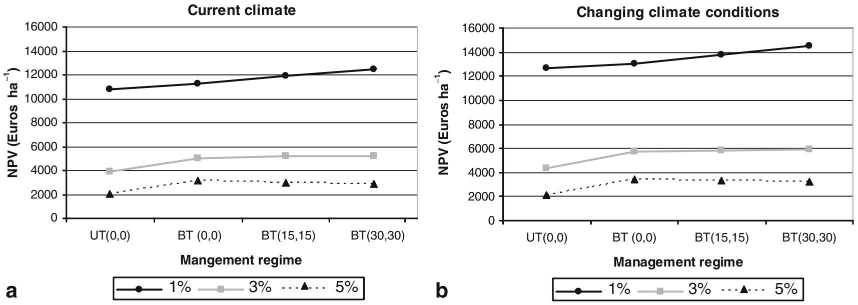


Fig. 3 Sensitivity of the net present value to the discounting rate under different management regimes and (A) the current and (B) changing climate scenarios when using a normal age class distributions and the species distribution 60% Norway spruce, 30% Scots pine, 10% silver birch (Case I)

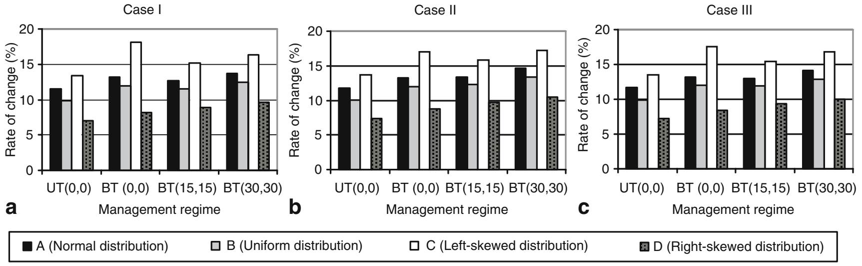


Fig. 4 Effect of climate change (HadCM2) on the NPV of harvested timber for different management regimes and for different species distributions, management scenarios and age class distributions. (A) Case I (60% Norway spruce, 30% Scots pine, and 10% Silver birch), (B) Case II (30% Norway spruce, 60% Scots pine, and 10% Silver birch), (C) Case III (45% Norway spruce, 45% Scots pine, and 10% Silver birch)

3.3 Effects of Forest Structure and Management on the Total Carbon Stocks in Forest Ecosystem over the 100-Year Simulation Period

Over the whole management unit and 100-year simulation period, the average C stocks per hectare were similar between different age class distributions. However, the highest C stock was observed for the forest with the right-skewed age class distribution and the smallest with the left-skewed distribution (Table 6). Taking as a baseline management scenario BT(0,0) with the current species proportions (Case I), the C stock was 104 Mg C ha^{-1} for right-skewed distribution, 102 Mg C ha^{-1} for uniform and normal distribution and 100 Mg C ha^{-1} for left-skewed distribution.

On the other hand, regardless of management and initial age class distribution, the species proportions in the unit affected the total C stock (Table 6). For example, for 100% Norway spruce, the C stock would have been 125 Mg C ha^{-1} compared to 70 Mg C ha^{-1} for 100% Scots pine under BT(0,0) with normal age class distribution. Regardless of the management and, climate scenarios, and initial age class distribution used, the total NPV of Norway spruce dominated structure (Case I) gave the highest amount of C stock. Thus, the average C stock of the unit was reduced when Scots pine area was increased and the Norway spruce was reduced (see Cases I-III in Table 6). The highest C stock in the forest ecosystem was calculated for the unthinned regime (UT(0,0)), about 45% higher than under the basic thinning regime (BT(0,0)). However, the increase in the thinning threshold and residual

Table 6 Carbon (C) in the ecosystem (Mg ha^{-1}, Ec, trees + soil) and in trees (Mg ha^{-1}, T), depending upon the age class distribution (A = normal distribution; B = uniform; C = skewed to the left; D = skewed to the right) and proportion of species in the unit (Case I resembles the current species distribution in the management unit (60% Norway spruce, 30% Scots pine, 10% Silver birch); Case II (30% Norway spruce, 60% Scots pine, 10% silver birch); Case III (45% Norway spruce, 45% Scots pine, 10% silver birch)

Management scenarios	Current climate								Climate change							
	A		B		C		D		A		B		C		D	
	Ec	T	Ec	T	Ec	T	Ec	T	Ec	T	Ec	T	Ec	T	Ec	T
Case I																
UT(0,0)	145	68	146	69	145	68	147	71	153	77	154	78	154	77	156	81
BT (0,0)	102	39	102	39	100	38	104	40	105	43	104	43	103	42	105	42
BT(15,15)	108	43	108	43	106	42	110	44	110	47	110	46	108	45	110	46
BT(30,30)	114	47	114	47	111	46	115	48	116	50	115	50	113	49	116	50
Case II																
UT(0,0)	118	56	119	56	119	56	118	57	125	63	125	64	125	63	126	65
BT(0,0)	86	35	86	35	84	34	87	35	88	38	88	38	87	38	89	38
BT(15,15)	91	39	91	39	89	38	92	39	93	42	93	42	91	41	94	42
BT(30,30)	96	42	96	42	94	41	97	42	98	46	98	45	97	45	99	46
Case III																
UT(0,0)	132	62	132	63	132	62	133	64	139	70	140	71	139	70	141	73
BT (0,0)	94	37	94	37	92	36	95	37	96	41	96	40	95	40	97	40
BT(15,15)	99	41	100	41	97	40	101	41	102	44	101	44	100	43	102	44
BT(30,30)	105	44	105	44	103	43	106	45	107	48	107	48	105	47	107	48

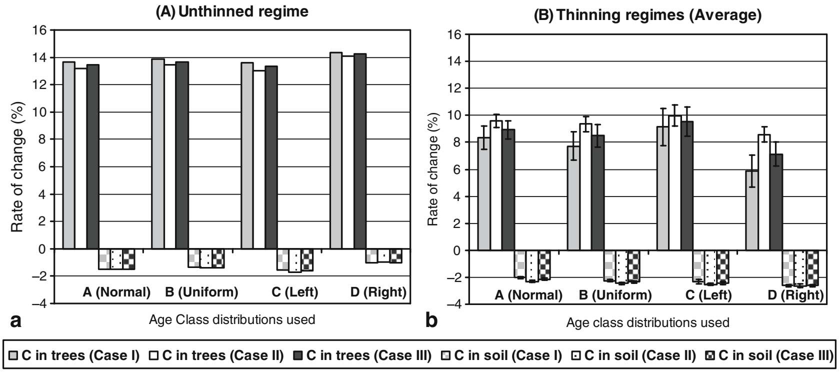


Fig. 5 Effect of climate change (HadCM2) on the C stocks (C in soil and C in trees) for different management regimes. (A) Unthinned regime and (B) Average for the thinning regimes, using different age classes distributions (A, B, C, D) and species distributions (Case I, II and III). Case I resembles current species distribution (60% Norway spruce, 30% Scots pine and 10% silver birch), Case II (30% Norway spruce, 60% Scots pine and 10% silver birch) and Case III (45% Norway spruce, 45% Scots pine and 10% silver birch)

basal are following thinning tended to increase the C stock (i.e., BT(15,15) and BT(30,30) versus BT(0,0), Table 6).

Under the changing climate, the effect of age class distribution, species proportions and the management scenario remained similar to current climate conditions. However, the climate change slightly increased C stock in the ecosystem in all the management regimes and all the forest structures compared (Table 6).

Management scenarios affected both C stocks in soil and in trees in a similar manner, but effects were higher on C in trees (Table 6). Climate change had different effects on the two C pools. An increase in temperature and precipitation increased the C stock in the trees, but the C stock was reduced in soils (Fig. 5). Also, climate change affected C stocks differently depending on the initial age class distribution and the species proportions, and especially for the thinning regimes (Fig. 5B) where differences were higher than for the unthinned regime (Fig. 5A). When the proportion of Scots pine dominated stands increased (Cases II-III), the effect of climate change on C in trees was higher than in the Case I where the unit was mainly dominated by Norway spruce.

3.4 Cost of Increasing Carbon Stocks in Forest Ecosystem over the 100-Year Simulation Period

Regardless of the climate scenario and forest structure used, the highest timber production over 100 years was found under the BT(30,30) thinning regime (Table 4), whereas the highest amount of C stocks was obtained under no thinning (UT(0,0))

(Table 6). Therefore, it was not possible to simultaneously achieve the maximum timber production and C stock in the forest ecosystem. However, it is possible to increase levels of both C stocks in the ecosystem and timber production over the currently used management scenario, BT(0,0).

Under the current climate and for the Norway spruce dominated structure (Case I), the increase in C stocks from BT(30,30) (maximum timber) to UT(0,0) (maximum C stocks), ranged between 31 Mg C ha^{-1} and 34 Mg C ha^{-1} depending upon the age class distribution (see Table 6). The selection of the UT(0,0) management regime instead of the BT(30,30) would increase C stocking at the potential marginal cost (**potMC**), ranging from 33.1€ Mg^{-1} for the right-skewed age class distribution to 42.8€ Mg^{-1} for the normal age class distribution when using discount rate of 3% (Table 7). Sensitivity analysis of the discount rate for the timber production showed that when the discount rate increased, the marginal cost for carbon sequestration decreased (Table 7). Moreover, changes in the species distribution in the management unit changed the results as shown in Table 7. For instance, when shifting from BT(30,30) to UT(0,0) regime, the difference of C stock is higher in the case of Norway spruce dominated forest structure (Case I) than in the Scots pine dominated structure (Case II). This difference had a great influence on the cost of C stocks. Cases II and III with an increase in the proportional area of Scots pine showed greater cost for C sequestration than for Case I.

Under climate change, the extra amount of C that can be stored when selecting UT(0,0) instead of BT(30,30) is higher than under current climate and the difference in NPV (loss of NPV) is also hither than under current climatic conditions. Thus the cost of enhancing C stock remained similar than under current climatic conditions and varied between 32.2€ Mg^{-1} and 42.8€ Mg^{-1} ($p = 3\%$) depending of the initial age class distribution analysed (see Table 7). The effects of age class distribution and share of species remained similar than under current climate conditions.

For the current marginal cost (**curMC),** BT(0,0) was assumed to be the business-as-usual management regime. Under current species distribution (Case I) and current climate, the results showed that shifting from current management BT(0,0) to the one that maximizes C stock (UT(0,0)) allowed the enhancement of the C sink from 42.8 to 44.5 Mg ha^{-1} (current climate) at a maximum price of 27.7€ Mg^{-1} ($p = 3\%$, Table 7). As in the case of potential marginal cost, whenever the share of Scots pine was increased in the unit (Cases II and III) the cost of C enhancement increased. Possible increases in carbon sinks were higher under climate change than under current climate due to the positive effect of increased temperature on the forest growth in boreal conditions.

Shifting from BT(0,0) to BT(30,30) allows to increase C stock in the ecosystem without any loss of NPV, which means that the thinning regime that maximises NPV increases the C stock in the forest compared to the current management rules applied and therefore there is not any cost. However, in absolute terms this is equivalent to only 10 to 12 Mg C ha^{-1} over 100 years depending on the original age class and species distributions.

Table 7 Potential[a] (potMC) and current[b] (curMC) marginal cost of Carbon sequestration by sink enhancement (€ Mg C^{-1}) depending on the share of the species in the unit (Case I, II and III) and on the age class distributions: A (normal distribution); B (uniform); C (skewed to the left) and D (skewed to the right). Discount rates used were 1%, 3% and 5%. The Case I resembles the original species distribution in the unit. Case I resembles the current share of species in the management unit (60% Norway spruce, 30% Scots pine, 10% silver birch), Case II (30% Norway spruce, 60% Scots pine, 10% silver birch), Case III (45% Norway spruce, 45% Scots pine, 10% silver birch)

Initial age class distribution	Current climate						Climate change					
	PotMC[a] (%)			CurMC[b] (%)			PotMC[a] (%)			CurMC[b] (%)		
	1	3	5	1	3	5	1	3	5	1	3	5
Case I												
A (normal)	54.1	42.8	28.7	10.5	27.7	26.1	49.6	42.8	28.2	7.1	29.0	25.3
B (uniform)	50.9	39.1	25.5	9.8	25.5	23.5	45.8	39.0	24.8	6.4	26.8	22.7
C (left)	54.4	38.9	23.5	14.3	23.7	20.3	51.3	40.2	23.8	12.6	27.7	20.7
D (right)	43.0	33.1	21.6	–0.1	22.3	21.5	36.2	32.2	20.2	–6.7	21.7	19.8
Case II												
A (normal)	95.9	59.8	38.2	28.0	37.2	32.6	99.2	62.1	38.5	27.2	38.8	32.1
B (uniform)	90.5	54.5	33.8	26.2	34.3	29.5	93.3	56.5	33.9	25.7	35.8	28.9
C (left)	94.6	54.0	31.4	31.7	32.5	25.5	100.0	57.8	32.7	33.1	36.4	26.2
D (right)	78.2	45.8	28.1	15.4	30.0	27.3	79.4	46.7	27.2	12.6	29.7	25.5
Case III												
A (normal)	71.7	49.9	32.7	18.1	31.8	29.0	70.2	50.8	32.5	15.8	33.2	28.3
B (uniform)	67.5	45.5	29.0	16.9	29.3	26.1	65.4	46.2	28.5	14.7	30.7	25.3
C (left)	71.4	45.3	26.8	21.9	27.5	22.6	71.6	47.6	27.5	21.5	31.5	23.1
D (right)	57.4	38.3	24.3	6.5	25.6	23.9	53.6	38.1	23.0	1.5	25.0	22.2

[a] Potential cost was derived from the differences in sequestered carbon and generated incomes from timber production of the best alternative for carbon sequestration and the best for timber production

[b] Current cost was derived from the differences in sequestered carbon and generated incomes from timber production of the best alternative for carbon sequestration and the current management used

4 Discussion

4.1 *Impacts of Forest Structure on Timber Production and Carbon Stocks*

Among the key issues addressed in managing forests is managing the distribution of tree species and stand age or tree size over the forest area in order to maximize long-term productivity. An understanding of the changes in forest production to the interactions between different changing environmental conditions and the initial forest structure and management is needed to evaluate forest management alternatives.

In the above context, the objective of this study was to assess how sensitive timber production and C stock are to the initial structure of a boreal forest landscape and management under the changing climatic conditions. We simulated four different age class distributions, two were extreme cases representing the forest areas dominated by very young stands (left-skewed age class distribution) versus mature stands (right-skewed age class distribution), and the other two represented normal and uniform age class distributions. We also simulated three different species distributions over the forest area, one resembling the actual state (dominance of Norway spruce) of forests in the management unit, used as a basis for comparisons, and the other two representing greater proportional areas of Scots pine dominated stands.

Regardless of the species proportions over the unit, the timber harvests were fairly balanced over time when the normal and uniform age class distributions were simulated. The left-skewed distribution (dominated by young stands) concentrated harvests on the latter years of the simulation period. Conversely, if the initial age class distribution representing mainly old stands was used, harvests were mainly concentrated at the early years of the projection period. In the latter case, the NPV of the timber harvest (discount rates 1%, 3% and 5%) was the highest because the value of the timber from harvests in the early years of simulation were less penalized by the discount rates than in other cases.

Regardless of the climate scenario and initial age class distribution, the maximum C stocks and the lowest NPV were observed when no thinning was done during the projection period. This was expected based on several previous studies (e.g., Dewar & Cannell, 1992; Karjalainen, 1996; Thornley & Cannell, 2000; Finér et al., 2003; Fürstenau et al., 2007). However, the C stocks in the forest ecosystem may be increased even in a managed forest, if high levels of tree stocking are retained prior to final harvest. In order to increase C stocks, it is also possible that the level of tree stocking may be further increased beyond those used in this study without increasing the mortality of trees. However, C stocks depend also on the tree species, stand structure, and properties of the site (Mäkipää et al., 1998, 1999; Vucetich et al., 2000; Pussinen et al., 2002).

Based on previous sensitivity analyses using the FINNFOR model to simulate changes in temperature and precipitation (Briceño-Elizondo et al., 2006a), in southern Finland the expected increase in temperature will enhance growth of Scots pine and silver birch, but expected changes in precipitation will affect the growth only marginally. Similarly in Norway spruce, growth may increase with temperature increases, but a concurrent reduction of precipitation will substantially reduce growth. As a result, Norway spruce may be replaced by Scots pine in the southern areas of the country due to decreased rainfall.

In our work, we showed how the predictions of timber yield and C stocks would vary depending of the proportional areas of species in the forest management unit. Regardless of the initial age class distribution, climate scenario and management used, the highest amounts of timber yield, NPV and also C stocks in the ecosystem where found when the management unit was 100% occupied by Norway spruce. Thus, if the current proportions in central Finland are changed resulting in increased Scots pine with a concurrent decrease in the area occupied by Norway spruce (Cases II and III), the values of timber yield, NPV and C stocks will decrease. In addition, the simulations showed that, regardless of the species distribution and management regime used, the initial age class distribution affected the increases (%) of timber yield, NPV and C stocks that were obtained from the changes in climate. As demonstrated here, our results can not be directly generalized to other areas without taking into account the structure (age class and species distributions) of the forest area at the beginning of the simulation. Thus, it is important to understand how different initial age class distributions and different species proportions over the management unit may affect forest productivity at landscape level.

We observed an increase in C stocks regardless of management scenario (unthinned or thinned scenarios) under changing climate due to an increased growth rate, as was found previously also by Karjalainen et al. (1999). This is also in line with the findings of Mäkipää et al. (1999) and Karjalainen et al. (2003), who indicated that a warmer climate in the boreal region could increase C stocks in forest vegetation by 10%. On the other hand, we found that the C in the soil decreased under the climate change compared to the current climate. This was due partly to higher temperatures, which enhanced the decomposition of SOM as claimed also previously in several studies (e.g. Grace, 2001, 2005; Karjalainen et al., 1999, 2003), but partly also due to the acceleration of soil respiration due to elevated temperatures (Peterjohn et al., 1994; Mäkipää et al., 1999). However, Thornley and Cannell (2001) suggested that C in soil may not always decrease in response to climatic warming, because the soil C may be more sensitive to the supply of suitable organic matter than to the changes in decomposition rate of SOM (Liski et al., 1999; Giardina & Ryan, 2000).

We found that, in managed forests, C sequestration may be increased by a change in management. The simulations also indicated that the business-as-usual recommendations might not be optimal for management under the climate change nor under the current climate with regards to C sequestration and timber production. This claim is supported by the findings that high stocking always increased the C sequestration compared to the stocking levels currently preferred. Thus, there is a clear need to further study how to adapt thinning schedules for the prevailing and changing climatic conditions to enhance C sequestration concurrently with timber production. However, the preference of C in the management induces opportunity costs for timber production especially if the management which results in the highest C stock is selected (UT(0,0)). When trying to increase the C stocks by changing the forest management, the cost of C sequestration would be higher for a management unit totally occupied by Scots pine than by Norway spruce. The larger the area covered by Scots pine (less Norway spruce), the larger the opportunity cost of C sequestration. In shifting from the base case management of BT(0,0) to management to maximize carbon (UT(0,0), differences in C stocks were smaller for Scots pine than for Norway spruce, but the difference of NPV were similar for the two species. Thus, the loss of NPV per tonne of carbon was much higher in Scots pine than in Norway spruce.

4.2 Possible Limitation of the Predictions

No model provides a perfect representation of ecosystem development. Forest ecosystems are complex and cannot be represented or described by the limited number of parameters in a model (Hasenauer et al., 2000). In addition, climate impact assessments using model predictions include uncertainties in modeling outputs from a variety of sources. In this context, sensitivity analysis techniques to simulate a range of possible parameter and data values and evaluate the effects on model output offer another approach. To evaluate the uncertainties of the FINNFOR outputs, sensitivity analyses of the outputs to changes in climatic parameters were presented in this study. In this context, it is also important to be aware of the possible changes in the physiological parameters of the plants due to global change. On the basis of a large experimental data set, Medlyn et al. (1999) showed that the photosynthesis sub-model presented by Von Caemerer and Farquhar (1981) used in FINNFOR, is likely to be adequate for predictions of long-term responses of forest to increases in atmospheric carbon dioxide (CO_2) concentrations. In addition, Medlyn et al. (2001) found no evidence of acclimation of stomatal conductance to elevated CO_2, but the maximum stomatal conductance in the Jarvis (1976) model used in FINNFOR could be expected to be reduced by 21%. On the other hand, Medlyn et al. (2001) indicated that the functions relating stomatal conductance to vapor pressure deficit (VPD), soil water potential and atmospheric CO_2 were generally unchanged in elevated CO_2 and hence do not need to be modified.

Another aspect that could change our results is the effect of climate on the frequency and magnitude of natural disturbances. Climate change is predicted to reduce water availability in some regions by decreasing precipitation and increasing the demand for water. Fire frequency and size of areas affected may also change over large areas of the world's forests (Spring et al., 2005), and the likelihood of other types of damage, for example breakage due to snow and ice, may change. These aspects were not taken into consideration during simulations. Also, we assumed that after the final harvest, stands would be planted with the species that were present prior to cutting. In practice, the choice of species in regeneration and subsequent management may be different from the previous species. Regarding C sequestration, this study focused on C stocks in the forest ecosystem; C stocks in wood products and possible effects of utilizing forest biomass as a fossil fuel substitute were not considered. Since C stocks in wood-products were excluded from the calculated opportunity costs of C sequestration by sink enhancement, actual opportunity costs of C sequestration would be lower than those calculated in the study.

As well as management for timber objectives, there is a need to explicitly address other goals such as carbon sequestration, biodiversity and recreation through multiple purpose forest management. In this study, the simulations were restricted to one management regime for the entire management unit. An optimization routine would be needed to simulate a mixture of management preferences over a forest area to better determine how forest management could be adapted to meet goals in the context of a changing climate.

5 Conclusions

This study demonstrated the use of a process-based model to predict forest development under different management regimes taking changing climatic conditions into account. Model results showed that forest growth and C stocks will generally increase under changing climatic conditions. Moreover, this study demonstrated the importance of analyzing changes in management regimes, particularly thinning thresholds and levels using simulation models, where there are no empirical data to inform decision makers. Simulations showed also the importance of taking into account the initial structure of the forest (in terms of age class distribution and proportions by species over the unit) when trying to predict future development. Results of this type can serve as a guide to forest managers and decision makers. In addition, they might be important for policy makers who might seek to promote policies that simultaneously enhance C sequestration and timber production. This study indicated that it may be possible to alter policy and management to enhance C sinks, without losses in monetary terms.

Acknowledgements This work was mainly funded through the Finnish Centre of Excellence Programme (2000–2005), under the Centre of Excellence for Forest Ecology and Management (Project no. 64308), co-coordinated by Prof. Seppo Kellomäki, University of Joensuu, Faculty of Forestry. Support provided by the Academy of Finland, the National Technology Agency (Tekes) and the University of Joensuu is acknowledged. Funding received from the EU through the project "Silvicultural Response Strategies to Climatic Change in Management of European Forests" under contract EVK2-2000-00723 (SilviStrat) is acknowledged. We express our gratitude to the partners of this Project for fruitful discussions and suggestions during the study design. Furthermore we thank Mr. Harri Strandman and Mr. Hannu Väisänen for helping with computer simulations. The authors would also like to thank Mr. Juha Hiltunen, Metsäkeskus Pohjois-Savo, for providing the forest stand data (X-forest-data) for the area. Last but not the least, we would like to thank Dr. Valerie LeMay as a reviewer for very helpful comments and suggestions for the manuscript.

References

Botkin, D.B. (1993). Forest Dynamics: An Ecological Model. Oxford University Press, Oxford, pp. 101–138.

Briceño-Elizondo, E., Garcia-Gonzalo, J., Peltola, H., Matala, J., Kellomäki, S. (2006a). Sensitivity of growth of Scots pine, Norway spruce and silver birch to climate change in boreal conditions, with needs for adaptive management. *Forest Ecology and Management 232*: 152–167.

Briceño-Elizondo, E., Garcia-Gonzalo, J., Peltola, H., Kellomäki, S., (2006b). Carbon stocks and timber yield in two boreal forest ecosystems under current and changing climatic conditions subjected to varying management regimes. *Environmental Science and Policy* 9: 237–252.

Cajander, A.K. (1949). Metsätyypit ja niiden merkitys. *Acta Forestalia Fennica 56*: 5–69 (in Finnish).

Carter, T., Bärlund, I., Fronzek, S., Kankaanpää, S., Kaivo-oja, J., Luukkanen, J., Wilenius, M., Tuomenvirta, H., Jylhä, K., Kahma,K., Johansson, M., Boman, H., Launiainen, J., Laurila, T., Lindfors, V., Tuovinen, J.-P., Aurela, M., Syri, S., Forsius, M., Karvosenoja, N. (2002). The

FINSKEN global change scenarios. In: Käyhkö, J., Talve, L. (eds.) Understanding the Global System, The Finnish Perspective. Painosalama, Turku, Finland, 232 p.

Chertov, O.G., Komarov, A.S. (1997). SOMM - a model of soil organic matter dynamics. *Ecological Modelling 94*: 177–189.

Dewar, R.C., Cannell, M.G.R. (1992). Carbon sequestration in the trees, soils and wood products of forest plantations: An analysis using UK examples. *Tree Physiology 11*: 49–71.

Erhard, M., Lindner, M., Cramer, W. (2001). Climate data. In: Kramer, K., Mohren, G.M.J, (eds.) Long-term Effects of Climate Change on Carbon Budgets of Forests in Europe. ALTERRA Report 194, ALTERRA, Wageningen, pp. 151–161.

Farquhar, G.D., von Caemmerer, S., Berry, J.A. (1980). A biochemical model of photosynthetic assimilation in leaves of C3 species. *Planta 149*: 67–90.

Finér, L., Mannerkoski, H., Piirainene, S., Starr, M. (2003). Carbon and nitrogen pools in an old-growth, Norway spruce mixed forest in eastern Finland and changes associated with clear-cuttings. *Forest Ecology and Management 174*: 51–63.

Finnish Statistical Yearbook of Forestry (2001). Finnish Forest Research Institute, Vantaa Research Unit (Finland).

Fürstenau, C., Badeck, F.W, Lasch, P., Lexer, M.J., Lindner, M., Mohr, P., Suckow, F. (2007). Multiple-use forest management in consideration of climate change and the interests of stakeholder groups. *European Journal of Forest Research* 126:225–239.

Garcia-Gonzalo, J., Peltola, H., Briceño, E., Kellomäki, S. (2007a) Effect of climate change and management on growth and timber yield in boreal forests, with economic implications: A case study, Ecological Modelling, 2009:220–234.

Garcia-Gonzalo, J., Peltola, H., Briceño, E., Kellomäki, S. (2007b). Changed thinning regimes may increase carbon stock under climate change: A case study from a Finnish boreal forest. *Climatic Change 81*: 431–454.

Garcia-Gonzalo, J., Peltola, H., Zubizarreta-Gerendiain, A., Kellomäki, S. (2007c). Impacts of forest landscape structure and management on timber production and carbon stocks in the boreal forest ecosystem under changing climate. *Forest Ecology and Management 241*: 243–257.

Giardina, C.P., Ryan, M.G. (2000). Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. *Nature 404*: 858–861.

Grace, J. (2001). Carbon Cycle. Edinburgh University Press, pp. 609–629.

Grace, J. (2005). Role of forest biomes in the global carbon balance. In: Griffiths, H., Jarvis, G.P. (eds.) The Carbon Balance of Forest Biomes. Bios Scientific, pp. 19–45.

Hasenauer, H., Burgmann, M., Lexer, M.J. (2000). Konzepte der Waldökösystemmodellirung (Concepts within Forest Ecosystem Modelling). *Central Blatt für das gesamte Forstwesen. 117*: 137–164.

Hynynen, J., Ojansuu, R., Hökkä, H., Salminen, H., Siipilehto, J., Haapala, P. (2002). Models for predicting stand development in MELA System. Research Papers 835. Finnish Forest Research Institute, 116 p.

IPCC (2001). In: (eds.) Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J., Dai, X., Maskell, K., Johnson, C.A. Climate Change: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York, 881 p.

Jarvis, P.G., (1976). The interpretation of the variation in leaf water potential and stomatal conductance found in canopies in the field. Phil. Trans. R. Soc. London, Ser. B273, 593–610.

Jarvis, P.G., Linder, S. (2000). Botany - constraints to growth of boreal forests. *Nature 405*: 904–905.

Jarvis, P.G., Ibrom, A., Linder, S. (2005). Carbon forestry – managing forests to conserve carbon. In: Griffiths, H., Jarvis, P.G. (eds.) The Carbon Balance of Forest Biomes. Taylor & Francis, London, pp. 331–349 (ISBN 1-8599-6214-9).

Karjalainen, T. (1996). Model computations on sequestration of carbon in managed forests and wood products under changing climatic conditions in Finland. *Journal of Environmental Management 47*: 311–328.

Karjalainen, T., Pussinen, A., Kellomäki, S., Mäakipää, R. (1999). Scenarios for the carbon balance of Finnish forests and wood products. *Environmental Science and Policy 2*: 165–175.

Karjalainen, T., Pussinen, A., Liski, J., Nabuurs, G.-J., Eggers, T., Lapveteläinen, T., Kaipainen, T. (2003). Scenario analysis of the impacts of forest management and climate change on the European forest sector carbon budget. *Forest Policy and Economics 5*: 141–155.

Kellomäki, S., Väisänen, H. (1997). Modelling the dynamics of the forest ecosystem for climate change studies in the boreal conditions. *Ecological Modelling 97*: 121–140.

Kellomäki, S., Karjalainen, T., Väisänen, H. (1997). More timber from boreal forests under changing climate? *Forest Ecology and Management 94*: 195–208.

Kellomäki, S., Wang, K.-Y., Lemettinen, M. (2000). Controlled environment chambers for investigating tree response to elevated CO_2 and temperature under boreal conditions. *Photosynthetica 38*: 69–81.

Kramer, K., Leinonen, I., Bartelink, H.H., Berbigier, P., Borghetti, M., Bernhofer, C., Cienciala, E., Dolman, A.J., Froer, O., Gracia, C.A., Granier, A., Grünwald, T., Hari, P., Jans, W., Kellomäki, S., Loustau, D., Magnani, F., Matteucci, G., Mohren, G.M.J., Moors, E., Nissinen, A., Peltola, H., Sabaté, S., Sanchez, A., Sontag, M., Valentini, R., Vesala, T. (2002). Evaluation of 6 process-based forest growth models based on eddy-covariance measurements of CO_2 and H_2O fluxes at 6 forest sites in Europe. *Global Change Biology 8*: 1–18.

Laasasenaho, J., Snellman, C.G. (1983). Männyn, kuusen ja koivun tilavuustaulukot. *Metsäntutkimuslaitoksen tiedonantoja* 113. Helsinki, 91 p. (in Finnish).

Lindner, M. (2000). Developing adaptive forest management strategies to cope with climate change. *Tree Physiology 20*: 299–307.

Liski, J., Ilvesniemi, H., Makela, A., Westman, C.J. (1999). CO2 emissions from soil in response to climatic warming are overestimated - the decomposition of old soil organic matter is tolerant of temperature. *Ambio 28*: 171–174.

Lloyd, J., Taylor, J.A. (1994). On the temperature dependence of soil respiration. *Functional Ecology 8*: 315–323.

Luo, Y.Q., Wan, S.Q., Hui, D.F., Wallace, L.L. (2001). Acclimatization of soil respiration to warming in a tall grass prairie. *Nature 413*: 622–625.

Mäkipää, R., Karjalainen, T., Pussinen, A., Kukkola, M., Kellomäki, S., Mälkönen, E. (1998). Applicability of a forest simulation model for estimating effects of nitrogen deposition on a forest ecosystem: Test of the validity of a gap-type model. *Forest Ecology and Management 108*(3): 239–250.

Mäkipää, R., Karjalainen, T., Pussinen, A., Kellomäki, S. (1999). Effects of climate change and nitrogen deposition on the carbon sequestration of a forest ecosystem in the boreal zone. *Canadian Journal of Forest Research 29*: 1490–1501.

Matala, J., Hynynen, J., Miina, J., Ojansuu, R., Peltola, H., Sievänen, R., Väisänen, H., Kellomäki, S. (2003). Comparison of a physiological model and a statistical model for prediction of growth and yield in boreal forests. *Ecological Modelling 161*: 95–116.

Medlyn, B.E., Badeck, F.-W., De Pury, D.G.G., Barton, C.V.M., Broadmeadow, M., Ceulemans, R., De Angelis, P., Forstreuter, M., Jach, M.E., Kellomäki, S., Laitat, E., Marek, M., Philippot, S., Rey, A., Strassemeyer, J., Laitinen, K., Liozon, R., Portier, B., Roberntz, P., Wang, K., Jstbid, P.G. (1999). Effects of elevated [CO_2] on photosynthesis in European forest species: A meta-analysis of model parameters. *Plant, Cell & Environment 22*(12): 1475–1495.

Medlyn, B.E., Barton, C.V.M., Broadmeadow, M.S.J., Ceulemans, R., De Angelis, P., Forstreuter, M., Freeman, M., Jackson, S.B., Kellomäki, S., Laitat, E., Rey, A., Roberntz, P., Sigurdsson, B.D., Strassemeyer, J., Wang, K., Curtis, P.S., Jarvis, P.G. (2001).Stomatal conductance of forest species after long-term exposure to elevated CO2 concentration: A synthesis. *New Phytologist 149*(2): 247–264.

Melillo, J.M., McGuire, A.D., Kicklighter, D.W., Moore III, B., Vorosmarty, C.J., Schloss, A.L. (1993). Global climate change and terrestrial net primary production. *Nature 363*: 234–239.

Nohrstedt, H.Ö. (2001). Responses of coniferous forest ecosystems on mineral soils to nutrient additions: A review of Swedish experiences. *Scandinavian Journal of Forest Research 16*: 555–573.

Olsson, P. (2006). Effects of Fertilisation on Rhizospheric and Heterotrophic Soil CO2 Efflux in Boreal Norway Spruce Stands. Licentiate thesis, Swedish University of Agricultural Sciences, Faculty of Forest Sciences, Department of Forest Ecology, Umeå. Stencilserie No. 113, 29 pp. ISSN 1104-1870, ISRN SLU-SEKOL-STL-113-SE.

Peterjohn, W.T., Melillo, J.M., Steudler, P.A., Newkirk, K.M., Bowles, F.P., Aber, J.D. (1994). Responses of trace gas fluxes and N availability to experimentally elevated soil temperatures. Ecological Applications *4*: 617–625.

Pussinen, A., Karjalainen, T., Mäkipää, R., Valsta L., Kellomäki, S. (2002). Forest carbon sequestration and harvests in Scots pine stand under different climate and nitrogen deposition scenarios, *Forest Ecology and Management 158*(1–3): 103–115.

Sabaté, S., Gracia, C.A., Sánchez, A. (2002). Likely effects of climate change on growth of Quercus ilex, Pinus halepensis, Pinus pinaster, Pinus sylvestris and Fagus sylvatica forests in the Mediterranean region. *Forest Ecology and Management 162*(1):23–37.

Sands, P.J., Battaglia, M., Mummery, D. (2000). Application of process-based models to forest management: Experience with ProMod, a simple plantation productivity model. *Tree Physiology 20*: 383–392.

Spring, D., Kennedy, J.O.S., Mac Nally, R. (2005). Optimal management of a forested catchment providing timber and carbon sequestration benefits: Climate change effects. *Global Environmental Change 15*: 281–292.

Strandman, H., Väisänen, H., Kellomäki, S. (1993). A procedure for generating synthetic weather records in conjunction of climatic scenario for modelling of ecological impacts of changing climate in boreal conditions. *Ecological Modelling 70*: 195–220.

Strömgren, M. (2001). Soil-Surface CO2 Flux and Growth in a Boreal Norway Spruce Stand. Doctoral thesis. Swedish University of Agricultural Sciences, Acta Universitatis Agriculturae Sueciae, Silvestria 220, 44 pp.

Thornley, J.H.M., Cannell, M.G.R. (2000). Managing forests for wood yield and carbon storage: A theoretical study. *Tree Physiology 20*: 477–484.

Thornley, J.H.M., Cannell, M.G.R. (2001). Managing soil carbon storage response to temperature: An hypothesis. *Annals of Botany 87*: 591–598.

Venäläinen, A., Tuomenvirta, H., Heikinheimo, M., Kellomäki, S., Peltola, H., Strandman, H., Väisänen, H. (2001). Impact of climate change on soil frost under snow cover in a forested landscape. *Climate Research 17*: 63–72.

Von Caemmerer, S., Farquhar, G.D. (1981). Some relationships between the biochemistry of photosynthesis and the gas exchange of leaves. *Planta 153*: 376–387.

Vucetich, J.A., Reed, D.D., Breymeyer, A., Degórski, M., Mroz, G.D., Solon, J., Roo-Zielinska, E., Noble, R. (2000). Carbon pools and ecosystem properties along a latitudinal gradient in northern Scots pine (*Pinus sylvestris*) forests. *Forest Ecology and Management 136*: 135–145.

Waring, R.H., Running, S.W. (1998). Forest Ecosystems: Analysis at Multiple Scales, 2nd edition. Academic, New York, 370 pp.

Yrjölä, T. (2002). Forest management guidelines and practises in Finland, Sweden and Norway. European Forest Institute. *Internal Report 11*: 1–46.

Part V
Case Studies

Carbon Sequestration in Mediterranean Pine Forests

M. del Río[1]*, I. Barbeito[2], A. Bravo-Oviedo[3], R. Calama[4], I. Cañellas[5], C. Herrero[6], and F. Bravo[7]

1 Introduction

Quantifying the carbon balance in forests is one of the main challenges if carbon fixation is to be considered amongst the objectives of forest management (Montero et al., 2005). Carbon is accumulated in forests through an increment in biomass, dead organic matter and soil carbon, and is released through respiration and decomposition. In the Mediterranean area, forest fires are also an important way of releasing CO_2 to the atmosphere. Due to the complexity of making a global evaluation of all these processes, studies of sub-processes are necessary to better understand the processes involved.

Carbon stored in vegetation is of great interest from a management point of view since, on one hand, carbon storage is easily modified through silvilcultural practices (e.g., rotation length, thinning, etc.), while, on the other hand, the amount of carbon affects the mean lifespan of wood products. Aboveground biomass is usually estimated from forest inventories through biomass equations and expansion factors

---

[1] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA Madrid, *delrio@inia.es

[2] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA Madrid, barbeito@inia.es

[3] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA Madrid, bravo@inia.es

[4] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA Madrid, calama@inia.es

[5] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA Madrid, canellas@inia.es

[6] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain, chdeaza@pvs.uva.es

[7] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain, fbravo@pvs.uva.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

at different spatial scales (e.g. Isaev et al., 1995; Schroeder et al., 1997; Fang & Wang, 2001; Barrio-Anta et al., 2006), whereas belowground biomass is often indirectly estimated from the aboveground biomass. Information for other biomass components such as litter, dead organic matter or soil carbon is less available, because these elements are more difficult to measure and in many cases are more spatially variable than other components. However, our knowledge with regard to these components has increased over the last decade (i.e. Isaev et al., 1995; Schlesinger & Andrews, 2000).

More research is required into the effects of forest management on the carbon (C) cycle so that C storage can be integrated into management strategies. In this respect, historical records are useful for analysing the effects of past management activities on C stocks and models can be developed to estimate future C stocks under different management alternatives (Kolari et al., 2004; Balboa-Murias et al., 2006). Furthermore, it is important to consider global change, since a modification in growth rates is expected for many species under forecasted changes in temperature and rainfall (Cao & Woodward, 1998; Schröter et al., 2005). Most empirical growth and yield models are based on historical data under different climatic conditions than those forecasted and are not able to account for these possible climatic changes (Pretzsch, 2002).

Because of expected climate changes, long-term forest management in some Mediterranean areas will require the development of locally adapted sustainable forest management practices in the form of new silvicultural strategies to improve the resilience of the ecosystem and thus enable the continued provision of goods and services, including C storage (Scarascia-Mugnozza et al., 2000). These new strategies should be based on historical analyses as well as on ecological knowledge in order to determine the extent to which the current condition of the forest is defined by traditional forest management practices.

In Mediterranean forests, species of the *Pinus* genus play a prominent role due to their widespread distribution and their ecological and socio-economical importance. Five species compose Mediterranean pine forests in Spain: *Pinus halenpensis* Mill. (aleppo pine), *P. nigra* Arn. (European black pine), *P. pinea* L. (Italian stone pine), *P. pinaster* Ait. (Maritime pine) and *P. sylvestris* L. (Scots pine). These species cover more than 4 million hectares as dominant species (Table 1), representing about 15% of the national forest surface and 37% of forests with canopy coverages above 20% (Cañellas et al., 2006).

Table 1 Area of pine forests in Spain by species according to the second National Forest Inventory (in thousands of hectares)

Species	Occuring as the dominant species	Co-dominant with other species	Total
Pinus halepensis	1,365	135	1,500
Pinus nigra	525	338	863
Pinus pinea	223	147	370
Pinus pinaster	1,058	626	1,684
Pinus sylvestris	840	370	1,210

Table 2 CO_2 Balance (million of Mg) for Mediterranean pine forests in Spain according to Montero et al. (2005)

Species	CO_2 (1990)	CO_2 sequestration (biomass growth/year)	CO_2 extractions (harvest/year)
Pinus halepensis	78.96	3.65	0.52
Pinus nigra	91.50	5.04	0.48
Pinus pinea	41.62	1.88	0.35
Pinus pinaster	159.47	8.41	4.20
Pinus sylvestris	165.10	9.41	0.91
Total	536.65	28.39	6.46

Because of the variability in forest typologies as well as in ecological and socio-economic conditions, the management objectives in Mediterranean pinewoods are diverse, although protection is a key function of many pine forests. In this respect, pine species are most frequently used in afforestation programmes and therefore, are of particular interest in terms of carbon sequestration.

According to Montero et al. (2005), carbon accumulation in Spanish pine forests was over 535 million of Mg in 1990, based on the second National Forest Inventory. The annual amount of biomass extracted as a result of pine harvesting is considerably less than that produced by the annual carbon sequestration through forest growth of around 22 million of Mg (Table 2). More than half of the total CO_2 stored in the pine forests is stored in *Pinus pinaster* and *P. sylvestris* forests.

In this chapter, estimates of carbon sequestration in Mediterranean pine forests from a number of studies and areas are presented along with associated information on how forest management influences this process. The estimates come from a number of sources including carbon stock estimates under different management plans using a chronosequence trial, under different thinning regimes or age structures using growth models, and from a model for estimating coarse woody debris.

2 Carbon Stocks over Time for Two Scots Pine Forests

Carbon stocks are known to vary greatly during stand development and depend upon the site and the type of management employed. Previous studies in Mediterranean Scots pine forests suggest that the total amount of biomass varies greatly between woodlands because of site differences (Gracia et al., 2000). Most carbon budget estimates are based on measurements from middle-aged or mature stands, and little data are available for different stages of forest succession (Kolari et al., 2004). In this section, we evaluate the relationship between carbon storage and different types of management over the length of the harvest rotation in Scots pine stands.

2.1 Dataset

A chronosequence trial was established in *Pinar de Valsain* and *Pinar de Navafria* managed Scots pine (*Pinus sylvestris* L.) forests located in the Central Mountain Range of Spain. In *Valsain*, a uniform shelterwood system has been applied, opening the stand gradually and allowing regeneration to take place naturally over a 40-year period. A moderate thinning regime was applied from the stem-exclusion stage onwards. In *Navafria*, soil preparation was generally required for natural regeneration to occur, although when this does not succeed, seedlings were planted. An intensive thinning regime was applied from the early stages.

Data were collected on eight 0.5 ha rectangular permanent research plots installed in *Valsain* and five in *Navafria* (Table 3). These plots cover all the current age classes in both forests: from 1–120 years in *Valsain*, where rotation length is 120 years, and from 1–100 years in *Navafria*, where rotation length is 100 years. Two additional plots were installed at *Valsain* in uneven stands at the upper and lower limits of the Scots pine elevation gradient where few silvicultural interventions are carried out. At the lower limit (1,300 m), Scots pine appears mixed with Pyrenean oak (*Quercus pyrenaica* Willd.) while at the upper limit (1,800 m), the pinewood is mixed with high mountain shrubs, mainly with *Juniperus communis* L. ssp *alpina*.

2.2 Biomass Equations and Carbon Estimations

Oven dry biomass for the whole tree was estimated as the sum of aboveground and belowground biomass components estimated using the equations developed by Montero et al. (2005) for Scots pine and for Pyrenean oak (Table 4).

From the data provided by CREAF (Ibañez et al., 2002), the estimated percent of biomass that is carbon for *P. sylvestris* and *Q. pyrenaica* trees was 50.9% and 47.5% respectively. Using these percents and the relationship between carbon and carbon dioxide molecules, a value for CO_2 accumulation could be estimated.

2.3 Carbon Stocks in the Two Forests

Differences between the two managed forests as well as between the even and uneven-aged stands can be seen in Table 5. Both forests follow a similar trend, reaching the maximum carbon dioxide fixation when the mean diameter for trees of more than 10 cm diameter outside bark at breast height (1.3 m above ground; DBH) reaches 30 cm and the density exceeds 500 trees per hectare. The stand is then gradually opened up to facilitate seedling establishment. The mean DBH at this point reached approximately 40 cm and the densities in both forests is reduced to approximately 300 trees per hectare with the resultant decrease in C.

Table 3 Stand level variables in the experimental plots at Valsain and Navafria: Plot size (m × m); Age class (years); N, number of trees per hectare; D, mean diameter outside bark at 1.3 m above ground (cm); BA, basal area (m^2/ha); H, mean height (m)

Forest	Plot	Plot size	Age class	N ≥10 cm	N <10 cm	D ≥10 cm	D <10 cm	BA	H ≥10 cm	H <10 cm
Valsain	V1	10 × 50	1–20	584	7,468	20.2	2.4	34.4	13.6	3.1
	V2	85 × 58.8	21–40	1,668	936	16.0	4.6	41.0	14.3	6.3
	V3	100 × 50	41–60	1,322	0	20.5	–	48.5	16.4	–
	V4	100 × 50	61–80	686	0	30.5	–	5.3..3	23.5	–
	V5	70.7 × 70.7	81–100	550	0	34.6	–	54	22.1	–
	V6	100 × 50	101–120	334	318	38.3	1.5	41.4	24.0	2.1
	V7	70.7 × 70.7	Uneven	608	964	18.8	1.6	25.2	11.5	2.4
	V8	100 × 50	Uneven	578	244	19.9	3.6	22.7	10.0	3.3
Navafria	N1	70.7 × 70.7	1–20	0	6,522	–	6.9	27.8	–	5.9
	N2	110 × 45.5	21–40	4,488	0	41.1	–	41.8	12.0	–
	N3	70.7 × 70.7	41–60	680	0	32.9	–	60.6	22	–
	N4	70.7 × 70.7	61–80	364	0	40.6	–	48.2	20.4	–
	N5	70.7 × 70.7	81–100	304	0	42.5	–	44.2	22.1	–

Table 4 Biomass equation parameters for different species and biomass fractions according to Montero et al. (2005)

		Biomass equation parameters ($\hat{B} = a \times DB\hat{H}^b$)	
Species	Biomass fraction	a	b
Scots pine	Bt	0.0844	2.4119
	Br	0.0109	2.6284
Pyrenean oak	Bt	0.0768	2.5345
	Br	0.0885	2.1335
Mediterranean maritime pine	Bt	0.0504	2.4964
	Bs	0.0327	2.5664
	Br	0.0215	2.3759
Stone pine	Bs	0.0352	2.5249
	Bb_2	0.0574	1.9394
	Bb_{2-7}	0.0171	2.3969
	Bb_7	0.0151	2.5999
	Bn	0.0184	2.1586
	Br	0.0183	2.4702

where Bt, Bs, Bb_2, Bb_{2-7}, Bb_7, Bn and Br are total aboveground, stem, branches under 2 cm diameter, branches between 2 and 7 cm diameter, branches above 7 cm diameter, needles and root biomass in kg respectively

Table 5 Aboveground, belowground, and total carbon dioxide fixation in Valsain and Navafria stands

Site	Age class (years)	Aboveground CO_2 (Mg/ha)	Belowground CO_2 (Mg/ha)	Total CO_2 (Mg/ha)
Valsain	1–20	276.91	77.00	353.91
	21–40	279.82	69.63	349.46
	41–60	359.72	93.01	452.73
	61–80	456.89	126.91	583.80
	81–100	479.69	135.49	615.18
	101–120	431.44	124.75	556.19
	Uneven upper limit	226.19	70.15	296.34
	Uneven, lower limit	182.31	49.66	231.97
Navafría	1–20	135.26	28.03	163.28
	21–40	263.51	62.31	325.82
	41–60	529.32	148.33	677.65
	61–80	451.85	131.25	583.10
	81–100	422.48	123.99	546.47

The lowest values for aboveground and belowground carbon dioxide fixation at *Valsain* were those for the uneven-aged plot situated at the upper limit of the pinewood. The only plot returning lower values was at *Navafria,* situated in a stand at an initial stage of development. The low CO_2 accumulation values for the uneven-aged plot situated at the upper limit of the pinewood are explained by the slow growth and low density in high-altitude environments. In the mixed oak and pine stand located at the lower limit of Scots pine distribution area, CO_2 accumulation values were found to be much lower than in almost all of the pine stands, except for age class 1–20 years, due to the smaller basal area per hectare.

The comparison of the two silvicultural systems revealed a similar carbon dioxide uptake over the length of the rotation, although the storage rate was gradual in Valsain and more abrupt in Navafria (Fig. 1). However, greater differences between the silvicultural systems applied in the two forests may become evident if the dead wood is taken into account (Montes & Cañellas, 2006). Previous studies suggest that lengthening the rotation length increases the carbon stocks in Scots pine forests (Kaipainen et al., 2004; Liski et al., 2004), although the results of the present study point towards a decrease in carbon stocks during the later stages.

The results for these forests cannot be readily compared with those of other studies because of differences in forest types, management systems and monitoring methodology. Another obstacle to such comparisons is that some studies only include parts of the carbon cycle. However, previous research concerning CO_2 values in the same geographical area (Bogino et al., 2007) also highlighted the difference between pure (482.63 Mg CO_2/ha) and mixed Scots pine stands (327.73 Mg CO_2/ha). When estimating the carbon stocks in Scots pine forests in Southern Finland, values of 216 Mg CO_2/ha for a 40-year old plot and 281 Mg CO_2/ha for a 75-year old plot were found for stands with a similar density (Kolari et al., 2004). These authors estimated the amount of carbon dioxide stored in soils to be similar to that stored in trees. Although we have no empirical evidence on which to base this statement, it may be reasonable to assume that the soils in the studied stands would be able to store an amount of carbon similar to that of the trees.

In addition to the disturbance caused by commercial harvesting, an increase in natural disturbances such as fires or plagues of insects might be expected as a result of climatic changes. The direct result of such disturbances is a decrease in the

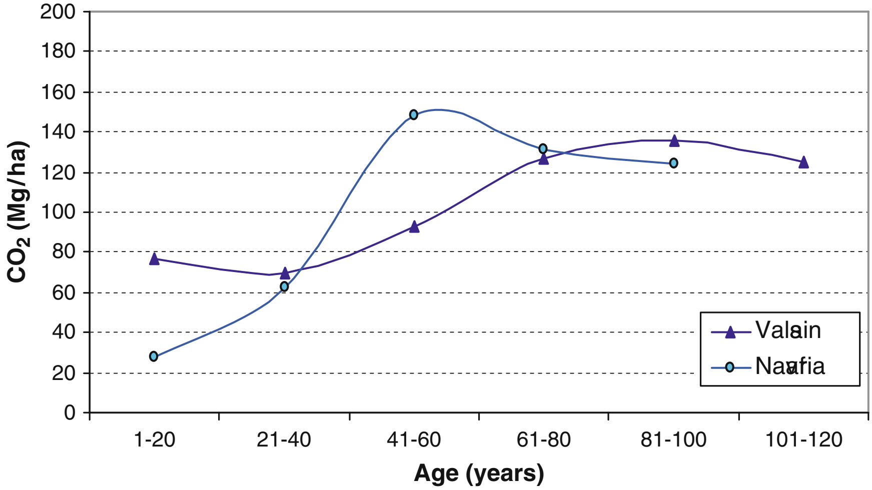


Fig. 1 Carbon stock of Scots Pine in Valsaín and Navafría

carbon stocks stored in vegetation, while the age-class distribution of the post-event forest tends towards the younger age classes which contain less carbon. Therefore, suitable strategies must be employed to mitigate these disturbances at stand level and to prevent forests from becoming sources of net carbon. Partial cover systems such as those currently employed in these forests, help retain biomass and therefore contribute positively towards the proposed objective. Reducing the delay in regeneration and avoiding slash burns would also help to strengthen the role of these forests as carbon sinks. Another option, which might be feasible in mixed stands such as the one included in this study, would be to promote those species which have a higher carbon storage potential (e.g., in this case, Scots pine). However, for appropriate management decisions to be made, further research is required into the effects of climate change on these carbon sinks and the variation in carbon budgets over the course of the rotation.

3 Management Alternatives and CO_2 Fixation in Mediterranean Maritime Pine

Forest management can help mitigate the effect of climate change through biomass accumulation. One way to increase CO_2 fixation in existing forests is to modify the strategies for harvesting and thinning. Dynamic growth models are useful tools for simulating different thinning alternatives and, when used in conjunction with biomass equations, these models allow us to determine carbon storage under different management alternatives (Kaipainen et al., 2004; Balboa-Murias et al., 2006). Montero et al. (2003) tested two silvicultural alternatives in *Pinus sylvestris* L. stands and found that the annual rate of carbon fixation was higher with more intensive management. Similar results were found by Bravo et al. (2008) when comparing different rotation lengths and site qualities in *Pinus sylvestris* L. and *Pinus pinaster* Ait. stands in Spain. The alternatives compared in most of the studies include some form of silvicultural intervention. In this study, we attempt to identify the most appropriate silvicultural regime for carbon fixation in Mediterranean maritime pine stands, using a growth model to compare different thinning alternatives, including an un-thinned scenario.

3.1 Growth and Yield Model

A dynamic growth and yield model was used to simulate the different silviculture alternatives at the stand level. The model consists of a compatible system of growth equations which estimates the current stand volume and predicts the future basal area and volume of the stand based on the site index and length of the projection. An equation for predicting top height growth (mean height of the 100 thickest trees per hectare) is also included in the model. Finally, a control function is used to simulate the response of the stand to thinning by estimating the quadratic mean diameter after thinning from the pre-treatment diameter and the thinning intensity. Parameterization was carried out using data from the permanent sample plot network belonging to the Forest Research Center (CIFOR-INIA) in Spain. For further details on the model see Bravo-Oviedo et al. (2004). In these simulations, natural mortality was considered to be null as this was controlled by thinnings, except for an un-thinned scenario.

3.2 Carbon Estimations

Carbon dioxide estimates were obtained using the equations described by Montero et al. (2005) (Table 4). The carbon in each fraction was calculated from oven-dry biomass by multiplying each value by 0.5 according to Kollmann (1959) and the Intergovernmental Panel on Climate Change (IPCC) recommendations (Penman et al., 2003). According to Guindeo et al. (1997), a mean basic density of 0.54 Mg/m^3 was assumed in Maritime pine stands in order to estimate the biomass of merchantable stems.

Changes in the carbon stock were computed for two (i.e., aboveground biomass and belowground biomass) of the five carbon pools usually considered in the land-use category "forest land remaining forest land" (Penman et al., 2003). Carbon stocks associated with dead wood, litter, and soil organic matter were not modelled.

3.3 Silvicultural Alternatives

The Mediterranean maritime pine stand simulated in this study approximates a real stand in the Central Mountain Range of Spain located on granite bedrock, with an annual precipitation of 1,400 mm and a drought period in summer (data from permanent plots). The initial stand characteristics for the simulation were: density of 1,500 stems per hectare, quadratic mean diameter (QMD) of 15.85 cm, basal area of 21.6 m^2/ha and site index of 21 m at 80 years according to the Bravo-Oviedo et al. (2004) site index curves for the species.

Using the model, eight thinning regimes were tested to evaluate potential carbon fixation (Table 6). All regimes included three thinning events, 10 years between thinning events, and all thinning events were from below, removing the smaller DBH trees. The age at which the first thinning was carried out varied among the alternative regimes from 20 to 30 years of age, and the percentage of basal area removed in each operation ranged from 20% (light thinning) to 35% (heavy thinning). Table 6 presents the characteristics of the regimes evaluated, separating the alternatives into those where a constant basal area was removed versus those where variable amounts of basal area were removed at each thinning event.

Table 6 Thinning regimes tested for carbon fixation. Three thinnings were applied for each alternative

Thinning regime group	Alternative	Age of first thinning	Basal area removed in each thinning (%)
Un-thinned	0	No thinning	
Constant basal area removed	1	20	35
	2	30	35
	3	20	20
	4	30	20
Variable basal area removed	5	20	35/20/20
	6	30	35/20/20
	7	20	20/20/35
	8	30	20/20/35

Table 7 Carbon dioxide fixed (Mg/ha) at rotation age (80 years old) and mean annual Carbon fixed (Mg/ha and year)

Thinning group regime	Alter native	QMD (cm)	CO_2 extracted in thinnings (1)	CO_2 fixed in stand (2)	Total CO_2 fixed (1) + (2)	Mean annual CO_2 fixed
No thinning	0	32.28	4.66[a]	688.1	692.8	8.7
Constant basal area removed	1	51.47	245.4	532.2	777.6	9.7
	2	41.52	321.7	409.2	730.9	9.1
	3	40.24	154.5	586.9	741.4	9.3
	4	37.17	207.0	507.4	714.4	8.9
Variable basal area removed	5	43.56	171.5	590.0	761.5	9.5
	6	37.91	237.4	495.3	732.7	8.8
	7	40.64	207.5	542.2	749.7	9.4
	8	35.17	264.8	453.8	518.6	8.9

[a] This value corresponds to mortality

For the scenario without thinning, natural mortality accounted for basal area reduction between 0.50% and 0.11%. These values were obtained by applying the mean mortality rate found for the species and a quadratic mean diameter for dead trees of 15 cm (Bravo-Oviedo et al., 2006). The best thinning regime was identified by comparing the total carbon fixed in each regime, taking into account all thinning events and final harvest, assuming a rotation length of 80 years. The mean annual carbon sequestration was calculated as the mean annual biomass increment.

3.4 Simulation Results

The main results of the simulations expressed in Mg of CO_2 fixed are shown in Table 7. The maximum annual sequestration was obtained with an early thinning (20 years), regardless of thinning regime group, and was slightly higher with at least one heavy thinning. The thinning regime which resulted in the greatest amount of fixed carbon was an early, heavy thinning (Alternative 1); the un-thinned alternative produced the lowest amounts of CO_2 fixed. A light, early thinning (Alternative 3) resulted in more carbon fixed than later thinning alternatives, although the results of the 30-year option improved with heavy initial thinning (Alternatives 2 and 6). The lowest amount of carbon fixed, apart from the un-thinned alternative, corresponded to Alternative 8, where the initial age at thinning was 30, and two light thinnings followed by a final heavy thinning were applied. These results suggest, firstly, that the first intervention should be carried out early and, secondly, that the response to heavy thinning is larger at younger ages. Furthermore, with a rotation length of this magnitude, intensive silviculture seems to be the most appropriate in terms of CO_2 fixation for the species.

The values presented in Table 7 were for the total biomass production including stem, branches, needles and roots. In all the alternatives, the amount of CO_2 sequestrated by the merchantable section of the main stem was always over 60% of the total (values not shown). This fact is important as wood products form another carbon sink, lengthening the duration of the carbon fixation and delaying its release to the atmosphere. Unfortunately, the model does not allow the diameter distribution to be calculated at the end of the rotation. However, Montero et al. (2003) found that intensive rather than extensive management in Scots pine stands leads to larger diameters. The different alternatives were compared by calculating the differences in the increment of carbon fixed for the merchantable section of the stem. Table 8 shows these comparisons where columns are the selected alternatives versus rows which are discarded alternatives. All the alternatives fixed from 2.85% to 12.47% more carbon than un-thinned stands. Alternative 1 was found to be best thinning regime of those included for the stand conditions simulated and as previously defined. Delaying the first thinning until the 30th year involved a loss of at least 4% for all thinning intensities. Similarly, when a light thinning was simulated, the loss in comparison to heavy thinning ranged from 2.5% to 4.7%. The application of any of the alternatives proposed was always preferable to no intervention at all in terms of the carbon fixed by the merchantable stem.

Table 8 Percentage of carbon fixed by stem. Selected alternatives in columns are compared with discarded alternatives in rows

		Selected alternative								
		1	2	3	4	5	6	7	8	0
Discarded alternative	1	–	−6.24	−4.72	−8.55	−1.99	−6.03	−3.64	−7.99	−11.1
	2	6.65	–	1.62	−2.47	4.53	0.22	2.77	−1.87	−5.17
	3	4.95	−1.59	–	−4.02	2.87	−1.37	1.14	−3.44	−6.68
	4	9.35	2.53	4.19	–	7.18	−1.37	5.38	0.61	−2.77
	5	2.03	−4.33	−2.79	−6.7	–	−4.12	−1.68	−6.13	−9.28
	6	6.41	−0.22	1.39	−2.69	4.29	–	2.54	−2.09	−9.28
	7	3.77	−2.7	−1.13	−5.1	1.77	−2.48	–	−4.52	−7.74
	8	8.69	1.91	3.56	−0.61	6.52	2.14	4.77	–	−3.36
	0	12.47	5.46	7.16	2.853	10.23	5.69	8.38	3.48	–

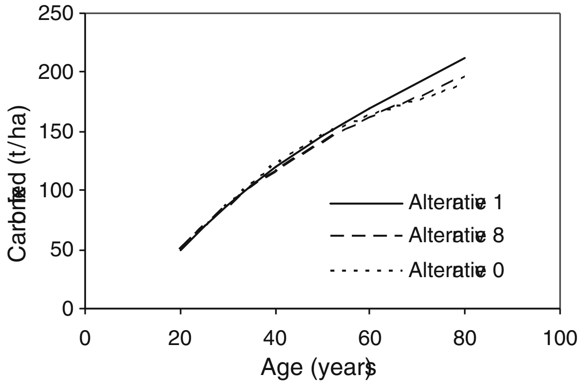


Fig. 2 Total carbon fixed in stems for three alternatives (see Table 6)

Figure 2 shows the total carbon fixed by the stem under the best thinning regime, the worst thinning regime, and the un-thinned stand over time (Alternatives 1, 8 and 0, respectively). These alternatives resulted in similar carbon storage up to an age of 50 years. At this age, just after the final thinning, Alternative 1 led to a larger amount of stored carbon than the other alternatives. More intense management appears to increase the carbon stored in the case of a long rotation period. However, if the rotation length was shortened, a less intensive alternative would probably lead to better results. For example Alternative 5 at age 50 years fixed 538 Mg CO_2 ha^{-1}, whereas Alternative 1 (more intense) at age 50 fixed 533.9 Mg CO_2 ha^{-1}.

Other studies have found the opposite tendency, with favourable carbon sequestration occurring when lower intensity intervention was applied (Balboa-Murias et al., 2006; Pohjola & Valsta, 2006). Losses in carbon sequestration are to be expected if intensive management exceeds the 'marginal thinning intensity' causing a loss in volume production (Hamilton, 1981).

Silvicultural experiences provide useful information for making decisions in relation to sustainable forest management. When such experiences are scarce or expensive, a modelling approach may help to identify the best alternative under different scenarios. Although the simulation described in this section did not include all factors involved over the course of the carbon cycle such as ground litter or carbon sequestration in the soil, the results confirm the importance of sustained silvicultural intervention over no intervention in terms of carbon fixation.

4 Carbon Sequestration in Even and Uneven Aged Stone Pine Stands

Stone pine stands have traditionally been managed as even-aged with low stocking densities, facilitating crown growth and increasing light resulting in greater cone production. However, uneven-aged stands also exist as a consequence of factors such as advanced recruitment, failure of natural regeneration, the impacts of animal grazing, and the preservation of older, large, cone-producing trees. Today, some of these stands are maintained and managed as uneven-aged to protect soils (especially, dune ecosystems), in landscaping, as recreational areas, or for fruit production. There are some local records on the management of these stands (Finat et al., 2000; Montero et al., 2003) as well as some studies comparing growth and cone yield in even and uneven-aged stone pine stands (Río et al., 2003; Calama et al., 2007). However, no studies on the inclusion of carbon sequestration were found. In this section we analyse the influence of age structure on carbon sequestration in stone pine stands. For this purpose, an individual-tree growth model and biomass equations were used.

4.1 Growth and Yield Model

The PINEA2 model is an integrated single-tree model, oriented towards multiple use management of stone pine stands. This model allows the growth and yield of a stand to be simulated under different management schedules and thinning regimes. The simulations are carried out in 5-year steps, defining the state of every tree within the stand at each stage of simulation. The model consists of three different modules: site quality, transition and state. The state module includes, among others, a taper function which allows end-use classification of timber volume according to size, a discriminant function for predicting probability of stem rot by *Phellinus pini*, and an equation for estimating annual cone production. The PINEA2 model was initially constructed and validated for even-aged stands. Given the single-tree character and the stochastic formulation of the functions included in the model, it was possible to calibrate it for a multi-aged complex forest structure. Further details regarding the PINEA2 model can be found in Calama et al. (2007).

4.2 Biomass Equations and Carbon Estimations

Tree biomass (by component) was estimated from DBH) using the biomass equations proposed by Montero et al. (2005) for stone pine (Table 4). Carbon for each component was calculated from oven-dry biomass by multiplying each value by 0.508 according to Ibáñez et al. (2002). Those carbon stocks associated with dead wood, litter, and soil organic matter were not modelled for this assessment.

4.3 Even and Uneven-Aged Alternatives

The growth and development of a 1 ha stand of stone pine located in the Northern Plateau of Spain was simulated under both even-aged and uneven-aged management structures over a 100-year period. In both cases, a site index of 17 m at a total age

Table 9 Proposed structure for uneven-aged stands of stone pine, rotation period 25 years

Age-class	Before selective felling (No. of trees/ha)	Removed in selective felling (No. of trees/ha)	After selective felling (No. of trees/ha)
0–25	110	60	50
25–50	90	55	35
50–75	35	15	20
75–100	20	10	10
100–125	10	5	5
125–150	5	5	0
TOTAL	270	150	120

of 100 years was assumed. Typical silvicultural programs applied in this region for multiple use management of stands with even and uneven-aged structures were compared:

1. In the even-aged stand, it was assumed that trees belong to the same age class, that the initial density is 500 stems per hectare at age 20, and that three thinnings are applied during the cycle: one systematic at age 30, reducing stand density to 350 stems per hectare; and two selective low thinning events at ages 45 and 60, reducing the density to 250 and 150 stems per hectare respectively. Regeneration cuts were carried out at 100 years.
2. In the uneven-aged structure, it was assumed that the stand displays the uneven equilibrium state (Table 9) proposed by Calama et al. (2005). Selective cuttings were applied every 25 years, removing non-vigorous and non fruit-producing trees in order to maintain a balanced representation of vigorous trees within the different age classes. A constant recruitment of 150 trees (DBH > 5 cm) per hectare and period was also assumed.

4.4 Carbon Sequestration in Even and Uneven-Aged Stands

The CO_2 fixed in stands of both age structures over the course of one rotation period (100 years) is presented in Fig. 3. The CO_2 fixed in the even-aged stand reached 386 Mg/ha by the end of the rotation period, while the simulated uneven-aged stand resulted in a maximum of 250 Mg/ha and a minimum of 150 Mg/ha. It should be pointed out that the uneven-aged stand maintained a constant amount of fixed CO_2 at around 150 Mg/ha which was never extracted from the forest.

If we take into consideration the annual growth of above and belowground biomass as well as annual cone production, the even-aged stand fixed 1.2 Mg/ha/year more CO_2 than the uneven-aged stand (Table 10), which means a difference of 120 Mg/ha over the 100 year period.

Biomass extractions totalled 540 Mg/ha of fixed CO_2 in the case of an even-aged structure, which was removed in three thinning interventions and at the final harvest at an age of 100 years (Fig. 4). Around 300 Mg/ha of this CO_2 was fixed in stems

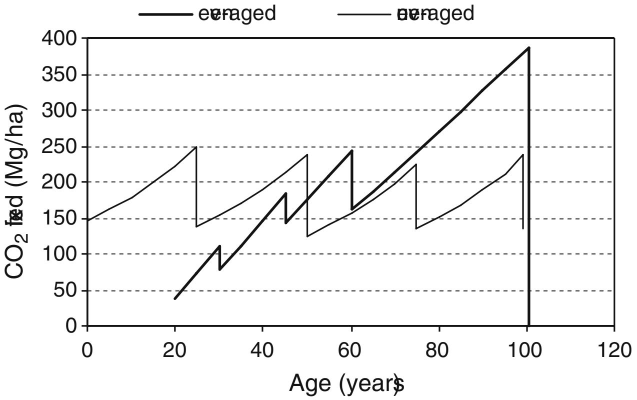


Fig. 3 CO_2 fixed by even and uneven-aged stands of stone pine over a period of 100 years

Table 10 Annual increments of above and belowground biomass, annual cone production and their equivalents in CO_2 fixed in even and uneven-aged stands of stone pine

		Aboveground	Belowground	Cones	Total
Biomass (kg/ha/year)	Even-aged	2,447.4	451.5	75.3	2,974.1
	Uneven-aged	1,832.4	348.7	105.4	2,286.4
CO_2 fixation (Mg/ha/year)	Even-aged	4.6	0.8	0.1	5.5
	Uneven-aged	3.4	0.7	0.2	4.3

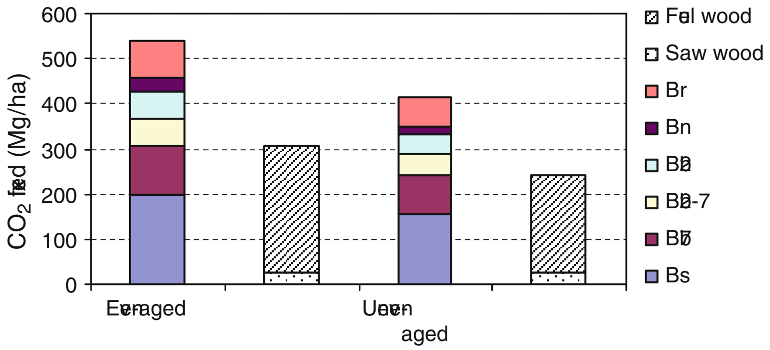


Fig. 4 CO_2 fixed in extracted biomass by component over the 100-year period in even and uneven-aged stands of stone pine. Bs, Bb_2, Bb_{2-7}, Bb_7, Bn and Br correspond to: stem, branches under 2 cm diameter, branches between 2 and 7 cm diameter, branches larger than 7 cm diameter, needles and roots. Saw wood includes stem timber free from rot affection obtained from sections with diameter >30 cm; fuel wood includes the rest of the stem timber as well as branches larger than 7 cm

and large branches, which would be used as pulpwood and saw timber, while the rest belongs to tree components which would remain in forest and decompose. In uneven-aged stands, four thinning treatments were applied over the 100 years (every 25 years), removing 416 Mg/ha of fixed CO_2, 243 Mg/ha of which would be destined for the timber industry (Fig. 4).

In terms of productivity, the uneven-aged structure favours cone production (Table 10), which is one of the main objectives in the management of stone pine forests. However, cone production does not contribute to CO_2 fixation, since most of the cones are collected each year to obtain edible pine nuts and any that remain are usually burned. With respect to timber production, the even-aged structure is more favourable, although in both cases, due to frequent stem rot, much of the timber production ends up as pulpwood or firewood. These two uses result in the short term return of carbon to the atmosphere.

To define the best age structure for a stand is a complex decision. Many forest products and services depend on age structure like wood production, CO_2 fixation, soil protection, habitat function, etc. and some of them are difficult to quantify and to compare between different age structures. The estimation of the CO_2 for even and uneven aged stands using growth models and biomass equations is a good option to include carbon sequestration in the decision making on selecting age structure.

5 Modelling Coarse Woody Debris in Pine Plantations

Dead wood plays an important role in the ecological processes of forest ecosystems. Although it is recognised that decaying logs and snags play an important role in forest biodiversity (Harmon et al., 1986; Esseen et al., 1992; McComn & Lindenmayer, 1999), little is known about dead wood dynamics in Mediterranean forests, where factors such as biodiversity conservation and carbon sequestration are of great importance.

The dynamics of this ecosystem is comprised of periods of undisturbed natural growth interrupted by natural disturbances produced by fire, wind, etc., or human intervention such as thinning or pruning. These disturbances, either small-scale gap perturbations or stand replacing catastrophic events, continuously replenish and create coarse woody debris (CWD) (Hansen et al., 1991). Under the paradigm of sustainable yield forest management, dead trees have been minimised to avoid pest problems and other hazards. Trees which die as a result of insect damage, disease or fire are commonly harvested immediately where economy and accessibility permit (DeBell et al., 1997). Today, the increasing importance given to both biodiversity and the carbon stocks in forests has led to the preservation and promotion of dead wood in managed forests. Forest and wildlife managers have suggested that five to ten snags per hectare are adequate to maintain the biodiversity (Hunter, 1990). Nevertheless, CWD and its relative contribution to the total ecosystem biomass vary greatly, depending on forest type, disturbance regime, topography and stand characteristics (Spies et al., 1988; Harmon & Chen, 1991).

In practice, snag and log dynamics are important to define the appropriate quantity, density, size (both diameter and height or length), distribution and state of decay of CWD in different site conditions and forest types (Hart, 1999; Woldendorp et al., 2004; Christensen et al, 2005; Stephens & Moghaddas, 2005). Studies which focus on modelling the abundance of snags and logs in Mediterranean type forest ecosystems are scarce (Montes & Cañellas, 2006). In this section, a snag/log abundance model is presented along with a carbon content equation for Mediterranean pine plantations composed of *Pinus sylvestris* L., *P. pinaster* Ait. and *P. nigra* Arn. in northern Spain.

5.1 Database

The study area, situated in the north of Spain constitutes a homogeneous transitional zone with altitudes ranging from 800 to 1,000 m asl., and an area of about 186,617 ha. The climate is Mediterranean with a slight Atlantic influence. Forests cover 59,471 ha (31.9% of total area) and are characterised by extensive stands of *Quercus pyrenaica* Wild., *Q. ilex* L. and *Q. faginea* Lam. As a result of an extensive pine plantation program carried out during the 1960s, *Pinus* stands cover 49.4% of the total forested surface of this area. The three main species composing the Pine plantations are *Pinus sylvestris* (23%), *P. nigra* (21%) and *P. pinaster* (5%). The soil in this region is mainly acidic, although there are also some limestone and neutral soils (Oria de Rueda et al., 1996).

Sixty six plots were installed in the study area in *Pinus* spp. planted stands (34 with a predominance of *Pinus sylvestris*, 24 of *P. nigra* and 8 of *P. pinaster*). The plots were composed of four subplots joined by two perpendicular transects. One of these subplots was a National Forest Inventory (NFI) plot with four concentric radii (Bravo and Montero, 2003) and the three other subplots were situated at the three vacant extremes of the two transects. An inventory of snags was performed in the four subplots while an inventory of logs was taken using the transects. The snags inventory was carried out by sampling 20 trees, spiralling out from the centre. Starting with the trees that were closest to the centre of the plot and moving progressively away, the condition of the trees was recorded, i.e. whether they were alive or dead. For large dead trees (DBH ≥ 7.5 cm), the variables recorded were: species, snag height, DBH, state of decomposition, presence of excavated cavities, and azimuth and distance to the centre of the plot. The log inventory was carried out in the two perpendicular transects of 50 m in length which joined the four subplots. Fallen dead trees with a diameter greater than 7.5 cm and a length greater than 1 m were considered logs. The following variables were measured: species, diameter at the point of intersection with the transect, length, state of decomposition and wildlife characteristics. Decomposition classes were followed the criteria by Sollins (1982).

The snag basal area (m^2) and log volume (m^3) were calculated for each plot. The individual basal area for each tree was totalled for each plot and the values

Table 11 Database characteristics used to develop the snag and log models for pine plantations in Northern Spain. N, trees per hectare; BA, basal area; QMD, quadratic mean diameter; BAsnags, basal area of snags; Vlogs, volume of logs

Variable	Mean	Minimum	Maximum	Standard deviation
N (trees/ha)	802.8	25.5	1,584.5	341.3
BA (m^2/ha)	23.2	5.6	39.3	8.2
QMD (cm)	22.21	13.17	58.27	6.29
BA_{snags} (m^2/ha)	1.7	0.2	14.9	3.6
V_{logs} (m^3/ha)	3.3	1.4	11.8	3.9

scaled up to give a basal area per hectare. The volume of logs was estimated through the equation (Warren & Olsen, 1964; Van Wagner, 1968): V = sum over all logs(($\pi^2 d_i^2$)/ 8L), where V: log volume (m^3/ha), d: diameter of each log (cm), L: length of the transect, which in this case was 100 m.

An intensive inventory of logs with a diameter greater than 1 cm was carried out in 32 out of the 67 study plots. This inventory was performed in the 10 m closest to the intersection of the two transects. The variables recorded for each log were: species, diameter at the interception point, length, weight, state of decomposition and wildlife characteristics. The mean diameter, mean length and mean weight were calculated by each plot. The total carbon content in samples of woody debris from each plot was measured through the instantaneous combustion of fragment samples in an oven at 550°C.

A description of the main stand variables is presented in Table 11. The following characteristics were also recorded for each plot: number of non-inventoried (i.e., sub-merchantable) stems, site conditions (soil texture, soil organic matter, pH, soil type, altitude, stoniness, slope, exposure and radiation), climate characteristics (rainfall, maximum, mean and minimum temperature, dry month rainfall obtained through a digital climatic atlas (Ninyerola et al., 2005)) and forest management history (harvests and thinning over the previous 15 years).

5.2 *Modelling Approach*

5.2.1 Two Step Regression Approach

A two step regression approach (Woollons, 1998; Álvarez-González et al., 2004; Bravo et al., 2007) was used to model the presence of CWD in pine plantations. In the first step, a logistic model was fitted to predict the probability of CWD presence, and in the second step, linear models were used to predict the basal area of snags and the volume of logs.

In the logistic model (Eq. 1), P is the probability of the presence of CWD, which is bound between 1 (presence) and 0 (absence), α is the intercept term, $\Sigma b_i X_i$, is the linear combination of parameters b_i and independent variables X_i, and e is the natural logarithm base.

$$P=(1+e^{-(\alpha+\Sigma b_i X_i)})^{-1} \quad (1)$$

Several predictor variables were used: QMD (cm); Ho, dominant height (m); N number of trees (trees per hectare); BA, basal area of stand (m^2/ha); BA_msp, basal area of dominant plot's species (m^2/ha); N_msp the number of trees of dominant species in the plot (trees per hectare); n, number of submerchantable stems (trees per hectare); S, slope (%); Alt, altitude (m); Exp, exposure; R, rainfall (mm), particularly, R_june, R_july, R_august for these months (mm); MaxT, maximum temperature (°C); MeanT, mean temperature, (°C); MinT, minimum temperature, (°C); and Rad, radiation (10 kJ/(m^2*day*μm)). Final logistic regression equations included only significant variables ($p < 0.05$). The goodness of fit was evaluated using the Hosmer and Lemeshow test (1989) and the Akaike Information Criterion (Zhang et al., 1997). PROC LOGISTIC of SAS (Version 8.1) was used (SAS, 2001). Receiver Operating Characteristic (ROC) curves for each model were used to compare the accuracy of different logistic regression models.

Linear models were used, as a second step, to predict the abundance of snags and logs (in terms of basal area and volume, respectively) in plots where the presence of CWD were predicted by using logistic model and a threshold value of 0.60. The linear model was:

$$\hat{y} = a_0 + \Sigma\, a_j X_j \quad (2)$$

Where $\hat{y}$ is estimated BA_{snag} or $Vol_{logs,}$; X_i are predictor variables, as in the logistic model; and a_0 and a_j are parameters to be estimated.

The precision of the two-step models were analyzed by comparing residual variance and dependent sample variance (Eq. 3) and by fitting a straight line between actual and predicted values (Huang et al., 2003), where slope and intercept should be equal to 1 and 0, respectively.

$$R^2 = 100^* \left(1 - \frac{S_e^2}{S_y^2}\right) \quad (3)$$

Where S_e^2 and S_y^2 are estimates of the residual sample variance and the dependent variable variance, respectively.

5.2.2 Carbon Content Model

A linear model was also used to estimate the carbon content of logs in the stands, where their presence was predicted using the logistic model. In this model, y is Clogs (carbon content in logs in %), and possible predictor variables (X_i) were: species (1 = Pine and 0 = Otherwise) and decomposition classes as dummy variables, mean diameter, mean length and mean weight of logs, stand variables (QMD, dominant height, number of trees per hectare, basal area per hectare, basal area per hectare for the dominant species in the plot, stems per hectare for the dominant species in the plot, number of small stems (trees < 7.5 cm DBH)), physiographic variables (slope, altitude, exposure) and climatic variables (rainfall, rainfall in the months of June, July, and August, maximum, mean and minimum temperature, radiation). Interactions between species and stand variables were also included as possible predictive variables.

The goodness of fit for the carbon model was assessed using the coefficient of multiple determination. Graphical and numeric analyses of the residuals (e_i) were performed to check assumptions of linearity, normality, and homogeneity of variance. Predictor variables were retained in the models if $p < 0.05$.

5.3 CWD Models

5.3.1 Snags and Logs Models

The final logistic model to predict the presence of CWD (Eq. 4) includes the following predictor variables: altitude, minimum temperature, clay and silt content, and a dummy variable indicating if a harvest operation was carried out in the last 5 years. The Akaike information criterion was 79.71, and the Hosmer and Lemeshow test (Pr > 0.5817) revealed no lack of fit. To determine the presence of CWD, a 0.60 threshold value was used, resulting in 68.2% of *Pinus* plots with CWD classified correctly (sensitivity equal to 51.5% and specificity equal to 84.8%). The area under the ROC curve was 0.7087.

$$\hat{P} = \left(1 + e^{-(-43.2715 + 2.5501 Alt + 0.3939 MinT + 4.6453\, ClayText + 2.4439 Silt\ Text + 2.5136 Harvesting)}\right)^{-1} \qquad (4)$$

where ClayText and SiltText are dummy variables for clay and silt textured soils, respectively, with both equal to 0 for sandy soils; and Harvesting is a dummy variable with 1 for harvest and 0 for no harvest. All other variables were previously defined.

The final linear models for snag basal area and log volume resulted in adjusted coefficients of determination of 17.47% for snags (Eq. 5) and 46.05% for logs (Eq. 6). The figure for snags increases with a decrease in the basal area of the main species in the stand, whereas the volume of logs increases with an increase in the basal area of the stand and when the dominant height decreases.

$$\hat{BA}_{snags} = 2.18462 - 0.0038 \times P\_june \qquad (5)$$

$$V\hat{o}l_{\log s} = 0.88211 + 0.20459 \times BA - 0.40278 \times Ho \qquad (6)$$

where $\hat{BA}_{snags}$ is the predicted basal area of snags in m^2/ha and $V\hat{o}l_{\log s}$ is the predicted volume of logs in m^3/ha. All other terms were previously defined.

The resulting two-step model achieved joint model accuracy equal to 39.25% and 62.75% for snags and logs, respectively. Graphic presentations of actual versus predicted values indicated that the joint model showed no lack of fit. The results of this empirical study may serve to understand more clearly the processes associated with an abundance of snags and logs.

5.3.2 Carbon Content Model

The final carbon content model obtained for logs (Eq. 7) resulted in a coefficient of multiple determination of 53.33% (adjusted coefficient of multiple determination was 47.50%). The carbon content of logs increased with an increase in the diameter of the logs, secondly, when the basal area of the stand increases, and finally, in the presence of small *Pinus* sp. trees. However, as the mean log diameter, basal area per hectare, and number of stems per hectare may be correlated variables, these coefficients should be interpreted with caution. The presence of small *Pinus* sp. trees is indicative of stands in early stages of development, where no management intervention has yet been carried out. For this study area, management appears to impact on the amount of carbon in logs.

$$\hat{C}_{\log s} = -54.57405 + 0.65940 \times m\log\, diam + 0.03457 \times BA - 0.00032779 \times Sp \times n \tag{7}$$

where *mlogdiam* is the mean diameter of the logs (cm); BA is the basal area of all live trees in the stand (m^2/ha), and Sp^*n is an interaction between the species dummy variable and the number of small stems (trees with diameter <7.5 cm) (trees per hectare).

Forest management decisions are based on information regarding current and future forest conditions. Therefore, it is often necessary to project changes in the system over time. The CWD estimation equations can be used to predict the presence of CWD and to quantify the quantity of snags and logs in pine plantations in northern Spain. The equations were developed for this area where *Pinus* plantations are managed for wood production, thinning is carried out over the course of the rotation, and the rotation is normally 40–50 years. However, the functions indicate that management practices affect both the amount of snags and logs, and also, the carbon percentages in logs.

6 Conclusions

We presented results of studies on how carbon sequestration changes over time, under different management regimes in Mediterranean pine forests. Chronosequences in Scots pine stands showed increasing carbon stocks with age, but differences were not found between the two compared forests. Growth models and biomass equations were used to estimate the amount of carbon fixed in forests under different management alternatives, such as different thinning regimes or different age structures. Intensive thinning regimes in Maritime pine forests appear to increase the total carbon fixed in stands; also, the application of any thinning regime always increased the amount of carbon stocks relative to no intervention. On the other hand, even-aged structures in stone pine stands seemed more favourable than uneven-aged, in terms of the carbon fixed. Since wood products provide a continuance in carbon storage, it is important to consider the size of the trees as well as the amount of carbon fixed in merchantable stems. The duration of carbon storage as wood products will depend on the type of product. More intensive thinning provides higher carbon fixed in the merchantable part of the stems and larger tree dimensions. Dead wood management (size, amount, density, decomposition status and their distribution throughout the forest) is currently one of the most important questions to be resolved for forest management, in the context of sustainability and biodiversity conservation. Accurate prediction and quantification of dead wood in the ecosystems is the first step to understanding CWD dynamics. The models presented for coarse woody debris allow us to quantify the biomass accumulated in this component, and therefore further our understanding of the carbon cycle in pine forests. Although using modelling approaches increases our knowledge, our understanding would be further improved by the use of growth models which include effect of climate change on growth estimates.

Acknowledgments This work has been funded through the University of Valladolid grant program and the following research projects: FORSEE, financed by European Union (INTERREG program); OT 03-002; CPE03-001-C5; AGL2001-1780, and AGL2004-07094-CO2-01 and -02/FOR.

References

Álvarez González, J.G., Castedo Dorado, F., Ruiz González, A.D., López Sánchez, C.A., von Gadow, K. (2004). A two-step mortality model for even-aged stands of *Pinus radiata* D. Don in Galicia (Northwestern Spain). *Annals of Forest Science*, *61*, 441–450.

Balboa-Murias, M.A., Rodríguez-Soalleiro, R., Merino, A., Alvarez-González, J.A. (2006). Temporal variations and distribution of carbon stocks in aboveground biomass of radiata pine and maritime pine pure stands under different silvicultural alternatives. *Forest Ecology and Management*, *237*, 29–38.

Barrio-Anta, M., Balboa-Murias, M.A., Castedo-Dorado, F., Diéguez-Aranda, U., Alvarez-González, J.A. (2006). An ecoregional model for estimating volume, biomass and carbon pools in maritime pine stands in Galicia (Northwestern Spain). *Forest Ecology and Management*, *223*, 24–34.

Bravo, F., Bravo-Oviedo, A., Díaz-Balterio, L. (2007). Carbon sequestration in Spanish Mediterranean forests under two management alternatives: a modeling approach. *European Journal of Forest Research 127*(3):225–234 DOI 10.1007/s10342–007.0198-y.

Bravo, F., Pando, V., Ordóñez, C., Lizarralde, I. (2007). Two-step ingrowth model for Scots pine (*Pinus sylvestris* l.) and Mediterranean maritime pine (*Pinus pinaster* Ait.) in Spain. *Internal Report*.

Bravo-Oviedo, A., Río, M., Montero, G. (2004). Site index curves and growth model for Mediterranean Maritime pine (*Pinus pinaster* Ait.) in Spain. *Forest Ecology and Management*, *201*, 187–197.

Bravo-Oviedo, A., Sterba, H., Río, M., Bravo, F. (2006). Competition-induced mortality for Mediterranean *Pinus pinaster* Ait. and *P. sylvestris* L. *Forest Ecology and Management*, 222, 88–98.

Calama, R., Finat, L., Gordo, F.J, Bachiller, A., Ruíz-Peinado, R., Montero, G. (2005). *Estudio comparativo de la producción de madera y piña en masas regulares e irregulares de Pinus pinea en la provincia de Valladolid*. IV Congreso Forestal Español. Mesa 3. SECF y Diputación General de Aragón, Zaragoza.

Calama, R. Sánchez-González, M., Montero G. 2007. Management oriented growth models for multifunctional Mediterranean forests: the case of stone pine (Pinus pinea L.) EFI proceedings *56*, 57–70.

Cañellas, I., Río, M., Montes, F., Vallejo, R. (2006). Forest and forestry in individual European Union countries, Spain. In: Arkuszewska A. et al. (Eds). *Forests and Forestry in European Union Countries*. The State Forests Information Centre and Forest Research Institute. Warsaw, pp. 417–432.

Cao, M.K, Woodward, F.I. (1998). Dynamic responses of terrestrial ecosystem carbon cycling to global climate change. *Nature*, *393*, 249–252.

Christensen, M., Hahn, K., Mountford, E.P., Standova, P., Rozenbergar, S., Diaci, J., Wijdeven, S., Meyer, P., Winter, S., Vrska, T. (2005). Dead wood in European beech (*Fagus sylvatica*) forest reserves. *Forest Ecology and Management*, *210*, 267–282.

DeBell, D.S., Curtis, R.C., Harrington, C.A., Tappeiner, J.C. (1997). Shaping Stand Development through silvicultural practices. In: A. Lathryn, Kohm, J.F. Franklin (Eds.) *Creating Forestry for the 21st Century. The Science of Ecosystem Management*. Island Press, Washington DC pp. 141–151.

Esseen, P.A., Ehnström, B., Ericson, L., Sjöberg, K. (1992). Boreal forest-the focal habitats of Fennoscandinavia. In: L. Hansson (Ed.) *Ecological Principles of Nature Conservation. Applications in Temperate and Boreal Environments*. Elsevier Applied Science, London, pp. 252–325.

Fang, J.Y., Wang, Z.M. (2001). Forest biomass estimation at regional and global levels, with special reference to China's forest biomass. *Ecology Research*, *16*, 587–592.

Finat, L., Campana, V., Seseña, A. (2000). La ordenación por entresaca en las masas de piñonero de la provincia de Valladolid. In I Simposio del pino piñonero (*Pinus pinea* L.), Valladolid. Junta de Castilla y León. pp. 147–157.

Gracia, C.A., Burriel, J.A., Mata, T., Vayreda, J. (2000). *Inventari Ecológic i Forestal de Catalunya*. Centre de Recerca Ecológica i Aplicacions Forestals.

Guindeo, A., García, L., Peraza, F., Arriaga, F., Kasner, C., Medina, G., de Palacios, P., Touza, M. (1997). *Especies de maderas para carpintería, construcción y mobiliario*. AITIM, Madrid.

Hansen, A.J., Spies, T.A., Swanson, F.J., Ohmann, J.L. (1991). Conserving biodiversity in managed forests. Lessons from natural forests. *BioScience*, *41*, 292–382.

Harmon, M.E., Chen (1991). Coarse woody debris in two oldgrowth ecosystems. Comparing a deciduous forest in China and a conifer forests in Oregon. *BioScience*, *41*, 604–610.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W., Cromack. K. Jr., Cummins, K.W. (1986). Ecology of coarse woody debris in temperate ecosystems. *Advances in Ecological Research*,*15*, 133–302.

Hart, S.C. (1999). Long-term decomposition of forest detritus in a Mediterranean type climate. In: *Abstract Presentations Book of the Blodgett Forest Symposium*. Center for Forestry, Berkeley, CA.

Hosmer, D.W., Lemeshow, S. (1989). *Applied Logistic Regression*. Wiley, New York.

Huang, S., Yang, Y., Wang, Y. (2003). A critical look at procedures for validating growth and yield models. In: A. Amaro, D. Reed, P. Soares (Eds.) *Modelling Forest Systems*. CABI Publishing, Wallingford, pp. 271–293.:.

Hunter, M.L., Jr. (1990). *Wildlife, Forest, and Forestry: Principles for Managing Forests for Biological Diversity*. Prentice-Hall, Englewood Cliffs, NJ.

Isaev, A., Korovin, G., Zamolodchikov, D., Utkin, A., Pryaznikov, A. (1995). Carbon stock and deposition in phytomass of the Russian forests. *Water Air Soil Pollution*, *82*, 247–256.

Kaipainen, T., Liski, J., Pussinen, A., Karjalainen, T. (2004). Managing carbon sinks by changing rotation length in European forests. *Environmental Science and Policy*, *7*, 205–219.

Kolari, P., Pumpanen, J., Rannik, U., Ilvesniemi, H., Hari, P., Berninger, F. (2004). Carbon balance of different aged Scots pine forests in Southern Finland. *Global Change Biology*, *10*, 1106–1119.

Kollmann, F. (1959). *Tecnología de la Madera y sus aplicaciones*. Tomo primero. IFIE, Madrid.

Liski, J., Pussinen, A., Pingoud, K., Mäkipää, R., Karjalainen, T. (2004). Which rotation length is favourable to carbon sequestration? *Canadian Journal of Forest Research*, *31*, 2004–2013.

McComn, W., Lindenmayer, D. (1999). Dying, dead, and down trees. In: L. Malcon, J.R. Hunter (Eds.) *Maintaining Biodiversity in Forest Ecosystems*. Cambridge University Press, Cambridge, 335–372 pp.

Montero, G., Cañadas, N., Yagüe, S., Bachiller, A., Calama, R., Garriga, E., Cañellas, I. (2003). Aportaciones al conocimiento de las masas de *Pinus pinea* L. en los Montes de Hoyo de Pinares (Ávila – España). *Revista Montes*, *73*, 30–40.

Montero, G., Río, M., Cañellas, I., Muñoz, M., Rojo, A. (2003). *Including CO2-Fixation in the Evaluation of Silvicultural Alternatives in Scots Pine Stands in Spain*. Decision Support for Multiple Purpose Forestry. Vienna, April 23–25.

Montero, G., Ruiz-Peinado, R., Muñoz, M. (2005). *Producción de Biomasa y fijación de* CO_2 *por los bosques españoles*. Monografías INIA, Serie Forestal, no. 13, 270 pp.

Montes, F., Cañellas, I. (2006). Modelling coarse woody debris dynamics in even-aged Scots pine forests. *Forest Ecology and Management*, *221*, 220–232.

Ninyerola, M., Pons, X., Rour, J.M. (2005). *Atlas climático digital de la Península Ibérica*. Metodología y aplicaciones en bioclimatología o geobotánica. ISBN 932560-8-7. Universidad Autónoma de Barcelona, Bellaterra.

Oria de Rueda, J.A., Díez, J., Rodríguez, M. (1996). *Guía de las plantas silvestres de Palencia*. Editorial Cálamo Palencia, 335 pp.

Penman, J., Gytarsky, M., Hiraishi, T., Drug, T., Kruger, D., Pipatti, R., Buendía, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F. (2003). *Good Practice Guidance for Land Use, Land-Use Change and Forestry*. Intergovernmental Panel on Climate Change (IPCC). Kanagawa (Japan).

Pohjola, J., Valsta, L. (2006). Carbon credits and management of Scots pine and Norway spruce stands in Finland. *Forest Policy and Economics*, *9*(7), 789–798.

Pretzsch, H. (2002). *Grundlagen der Waldwachstumsforschung*.Parey Verlag, München.

Río, M., Calama, R., Montes, F., Montero, G. (2003). Influence of competition and structural diversity on basal area growth in uneven-aged stands of stone pine (Pinus pinea L.) in Spain. *Uneven-Aged Forest Management: Alternative Forms, Practices and Constraints*. IUFRO-Congress, Helsinki (Finlandia), June 2003.

Scarascia-Mugnozza, G., Oswald, H., Piussi, P., Radoglou, K. (2000). Forest of the Mediterranean region: Gaps in knowledge and research needs. *Forest Ecology and Management*, *132*, 97–109.

Schlesinger, W.H., Andrews, J.A. (2000). Soil respiration and the global carbon cycle. *Biogeochemistry*, *48*, 7–20.

Schroeder, P., Brown, S., Mo, J., Birdsey, R., Cieszewski, C. (1997). Biomass estimation for temperate broadleaf forests of the United States using inventory data. *Forest Science*, *43*, 424–434.

Schröter, D., Cramer, W., Leemans, R., Prentice, C., Araújo, M., Arnell, N., Bondeau, A., Bugmann, H., Carter, T., Gracia, C., de la Vega-Leinert, A., Erhard, M., Ewert, F., Glendining, M., House, J., Kankaanpää, S., Klein, R., Lavorel, S., Lindner, M., Metzger, M., Meyer, J., Mitchell, T., Reginster, I., Rounsevell, M., Sabaté, S., Sitch, S., Smith, B., Smith, J., Smith, P., Sykes, M., Thonicke, K., Thuiller, W., Tuck, G., Zaehle, S., Zierl, B. (2005). Ecosystem service supply and vulnerability to global change in Europe. *Science*, *310*(5752), 1333–1337.

Spies, T.A., Franklin, J.F., Thomas, T.B. (1988). Coarse woody debris in douglas-fir forests of western Oregon and Washington. *Ecology*, *69*, 1689–1702.

Stephens, S.L., Moghaddas, J.J. (2005). Fuel treatment effects on snags and coarse woody debris in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, *214*, 53–64.

Van Wagner, C.E. (1968). The line-intersect method in forest fuel sampling. *Forest Science*, *14*, 20–26.

Warren, W.G., Olsen, P.F. (1964). A line-intersect technique for assessing logging waste. *Forest Science*, *10*, 267–276.

Woldendorp, G., Keenan, R.J., Barry, S., Spencer, R.D. (2004). Analysis of samplig methods for coarsy woody debris. *Forest Ecology and Management*, *198*, 133–148.

Woollons, R.C. (1998). Even-aged stand mortality estimation through a two-step regression process. *Forest Ecology Management*, *105*, 189–195.

Zhang, S., Amateis, R.L., Burkhart, H.E. (1997). Constraining individual tree diameter increment and survival models for loblolly pine plantations. *Forest Science*, *43*, 414–423.

Carbon Sequestration of Ponderosa Pine Plantations in Northwestern Patagonia

P. Laclau[1]*, E. Andenmatten[1], F. J. Letourneau[1], and G. Loguercio[2]

1 Introduction

Forest plantations are yet at a starting point in Argentine Patagonia. Since the first pine plantations were settled in the early 1970s, landowners and local governments have been indeed interested in forestry as a means to diversifying the dominant cattle monoculture. However, climate and soil (site) attributes-driving factors of the long term rotation periods and environmental risk-associated with current local wood prices, delayed the forest development. In this context, carbon trade arises as a new market service that could compensate for these drawbacks. This study case describes the physical environment were forests plantations evolve (subtitle 2), the land potential and carbon baseline for afforestation (subtitles 3 and 4) and analyses the stand growth and CO_2 capture, providing orientative data and empirical models of use to account for sequestered carbon (subtitle 5). Also, regional scenarios of carbon sequestration by forests at stand, watershed and regional levels are presented (subtitle 6).

The first experimental forest plots – started about 80 years ago –, showed a good adaptation and growth of ponderosa pine to local conditions (Tortorelli, 1955; Dimitri, 1972). Nowadays, afforestation for industry purpose give account for approximately 75,000 ha plantations, established at moderate rates, and at the expense of cattle substitution (Laclau et al., 2003; Loguercio & Deccechis, 2006). Although the usual management is industry-directed with dense plantations, forest research have recently focused on the environmental benefits (and costs) of forestry, including biomass allocation, impacts on biodiversity, plant invasions, and water balance (Buduba et al., 2002; Gyenge et al., 2002, 2003; Laclau, 2003; Corley et al., 2005; Rusch & Schlichter, 2005; Rusch et al., 2005a, b; Loguercio et al., 2005; Sarasola et al., submitted). Also, the arising of market rules to accomplish carbon

---

[1] Instituto Nacional de Tecnología Agropecuaria (INTA), Estación Experimental Agropecuaria Bariloche, CC 277, (8400) San Carlos de Bariloche, Río Negro, Argentina,
*placlau@bariloche.inta.gov.ar

[2] Centro de Investigación y Experimentación Andino Patagónico CIEFAP., Ruta 259 km4, (9200) Esquel, Chubut, Argentina

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

emissions reductions provided a frame of economic interest to study the carbon sequestration by forests, as a mean for climate change mitigation (Laclau, 2003; Loguercio et al., 2005). In the region, the afforestation potential can be supported by the following features:

- Availability of extensive and suitable lands for planting
- Relatively low carbon baseline
- Sound growth of ponderosa pine
- Adequate economic and social frame for forest development

In spite of its extensive area under fair conditions for plant growth, the potential for carbon sequestration by forest plantations still remains scarcely known, making necessary to describe the main physical driving factors involved, along with the most recent local findings about stand growth and carbon estimation. The ultimate objective of this chapter is to bring a comprehensional view of the capacity of the region to accomplish with climate mitigation throughout forest plantation projects.

2 Geographical Context

Argentine Patagonia is characterized for a high physiognomic, soil and climate variability (Oesterheld et al., 1998). Its geographic boundaries are about 36° and 55° S, and 65° and 73° W (Fig. 1). The regional climate is cold temperate, with a

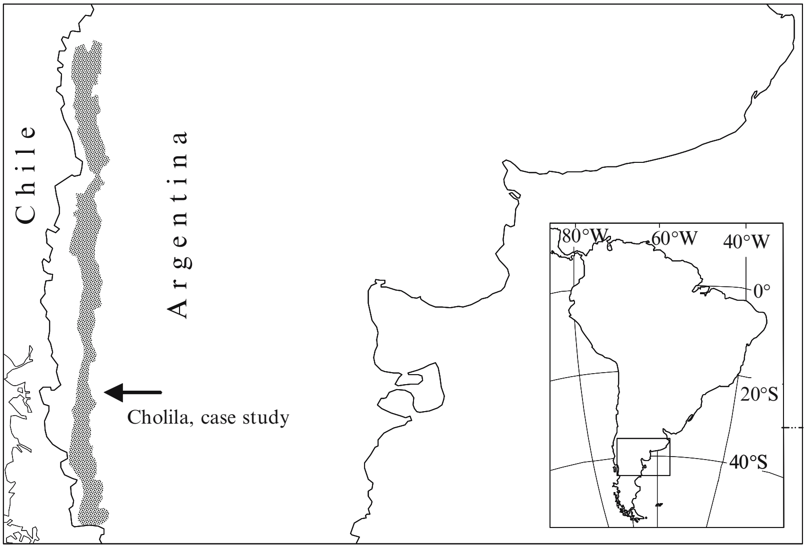


Fig. 1 Map location of the subandean region of Northwest Patagonia, suitable for ponderosa pine afforestation (dotted area) and location of the case study (Section 5.2)

decreasing moisture gradient from the Andean cordillera to the Atlantic Ocean (Cordon et al., 1993), and another north-south temperature-decreasing gradient, locally modified by the relief, the altitude and the neighbourhood to the sea (Oesterheld et al., 1998; Paruelo et al., 1998). At high elevations in the mountains, the climate is cold, frost-frequent along year and snow-frequent in fall and winter. The regional soils derive from volcanic ash, unweathered volcanic materials, and fluvial or colluvial sands and loans (Etchevehere, *in:* Dimitri, 1972). The Andean orographic barrier – which causes a steep decrease of the precipitation in a short distance – promoted a different soil genesis along the longitude gradient. The ashes, carried over by the prevalent Pacific winds, deposited over the rock layers, or mixed with glacial till. To the west, the dominant soils (Andosoils) developed under udic regime (3,000–1,000 mm *per* year), characterise by the presence of allophanes, weakly weathered pumicite and pyroclastic vitric materials (Candan et al., 2003) and for a high water retention capacity (Shoji et al., 1993). To the east, along a transition shorter than 100 km wide, the soils are classified as Mollisols, developed under xeric regime (<700–800 mm *per* year) (Colmet-Dâage, 1992; Candan et al., 2003).

Patagonia comprises two strong climate and topographic contrasting landscapes: the Andean cordillera and the Patagonian plateau. The plateau is characterized by aridity and a harsh climate (Paruelo et al., 1998). In the wet or mesic environments of montane valleys and hillsides, the best soils and conditions for vegetation growth are found (Dimitri, 1972; Paruelo et al., 1998). The temperate southern forests of *Nothofagus* and conifers grow there in wet forests and associate environments of the Valdivian Eco-region (Dinerstein et al., 1995; Armesto et al., 1997). To the east, in a transition characterized by steep rainfall decrease and relief attenuation (De Fina, *in:* Dimitri, 1972), lands suitable for afforestation are found (Fig. 1). This vaguely defined ecotone zone – limited by an annual rainfall range from about 1,200 to 500 mm –, includes some mountain ranges lower than the Andean cordillera and the most important collector rivers of the montane runoff, showing a grass and shrub steppe physiognomy, with patch inclusions of xerophytic native forests, grass meadows, and gallery woody thickets.

Fire and herbivory have been pointed out as the main driving variables of the structure and dynamics of the ecotone vegetation (Veblen et al., 1997; Kitzberger et al., 1997; Golluscio et al., 1998). Some natural causes of burnings are the pronounced water deficit during the hot summers, lightnings or volcanic events, and the accumulation of grass biomass as fuel. From the early colonization in the last century – and also before, due to the indigenous people activities –, human intervention became an important burning factor, depleting forests and steppe (Veblen & Lorenz, 1988). Early in the 20th century, Rothkugel (1916) highlighted the increasing frecuency of fires due to land conversion to pasture for cattle breeding. Also, sheep and cattle use, characterized by high animal stocking, inadequate paddock layout, and lack of pasture rotation (Golluscio et al., 1998), burst into soil erosion processes and plant cover degradation, some cases severe.

3 Forest Lands and Site Quality

To assess the suitability of land for conifer afforestation, Mendía and Irisarri (1986), Ferrer et al. (1990); Irisarri and Mendía (1991), Irisarri et al. (1997) developed an index that combined the physical soil features: *texture, drainage, effective depth*, and *water storage capacity of plant-available soil water*. These variables were weighed according to relevant conditions for tree growth; *e.g.*, if the *water storage capacity* in the soil profile was more than 120 mm, the highest score was assigned (ten points). Instead, if it was less than 30 mm, it was considered *limiting*, with a score of zero (0). The sum of all the variables scores for each soil type was allocated to a conversion table that qualified the land into the following classes and resulting land distribution: *Very suitable* (7% of the forest lands), *Suitable* (22%), *Moderately suitable* (54%), *Poorly suitable* (2%), *Marginally suitable* (14%), and *Unsuitable* (matrix of non-forest lands). The estimated area for such classes was recently revised by Loguercio and Deccechis (2006) using the same sources and data of the national native forests inventory (SAyDS, 2005). According to this, the potential afforestation area for ponderosa pine and other conifers is about 2 M ha, with an estimated volume grow ranging from 10 to 30 m^3 ha $^-$ year^{-1}.

Other studies addressing forest productivity and site quality related soil properties, climate and relief to *water storage* and *evapotranspiration loss* (Broquen et al., 2003; Andenmatten et al., 2002; Davel & Ortega, 2003; Loguercio et al., 2004). At the regional level, Andenmatten et al. (2002) studied the influence of various environmental variables on the site index (SI).[1] Through multivariate analysis techniques, they found that the variance of the SI was strongly explained – among other edaphic and climatic factors – by *water storage capacity* and *plant available water*. Also, they found that both thick (sandy to loamy-sand) and fine (clay) textures negatively correlated to dominant tree height. Instead, sandy-loam to silt-loam textures were associated to the higher SI values. As for example, low precipitation sites (<450 mm *per* year), western aspect, stony soil surface, "A" horizon depth of 18 cm, and effective depth of the soil profile of <80 cm, was linked to a $SI_{20} < 10$ m, while sites of >900 mm *per* year, eastern aspect, loamy soil texture, stones absent, "A" horizon of 24 cm, and effective soil depth of 100 cm, had a $SI_{20} > 20$ m.

4 Baseline

The dominant physiognomic types of the land that can be subject to afforestation projects are grass and shrub steppes, including patches of native forests, woody thickets, grass meadows and waste lands. All of these communities are somewhat disturbed by cattle or burnings. The breeding of cattle is the main land use in mesic

---

[1] *The SI is a species specific index defined as the mean height of the 100 thickest trees per hectare (dominant tree height, Assmann 1970), at 20 years age at breast height (abh) (Andenmatten and Letourneau, 1997).*

sites, although 40 years before, sheep was also important. Nowadays, goat and sheep farming is a major land use in the more xeric areas of the region. Pine plantations often replace these steppe environments, so as they can be considered a regional baseline for the C pool (Laclau, 2003). The baseline characteristics are:

- Short vegetation (usually below 1 m) with a cover range between 40% and 80%, dominated by tussock grasses and shrubs (Fig. 2)
- Plant species functionally or morphologically adapted to water stress, or herbivory resistant, with high C/N tissues, and shrubs of high root/shoot ratios
- Continuous or recurrent grazing by domestic and wild animals
- Greenhouse-gas emissions, due to rumen activity and natural or human-induced fires
- Slight to moderate soil erosion processes, related to cattle farming and burnings, facilitated by the topography and the relatively massive soil structure

Austin and Sala (2002), found that in the ecotone (100–800 mm annual rainfall) between the arid shrub-steppe and the deciduous forest the aerial net primary productivity (ANPP) of the vegetation is mainly controlled by the precipitation, highlighting the limiting effect of the available soil water. On the other hand, Bertiller and Bisigato (1998), characterized the changes in the patagonian steppe under increasing rates of grazing, following the state-and-transition model approach (Paruelo et al., 1993). The progressive degradation due to overgrazing is distinguished by a plant cover decrease, the replacement of species or functional groups, and the unchaining of soil erosion processes (Bertiller & Bisigato, 1998), trending

Fig. 2 Sample plot (2 × 1 m) for the biomass assessment on a grass-shrub steppe of Mulinum spinosum, Acaena splendens, Festuca pallescens and other herb species, in Meliquina, Neuquén (Photo: G. Stecher)

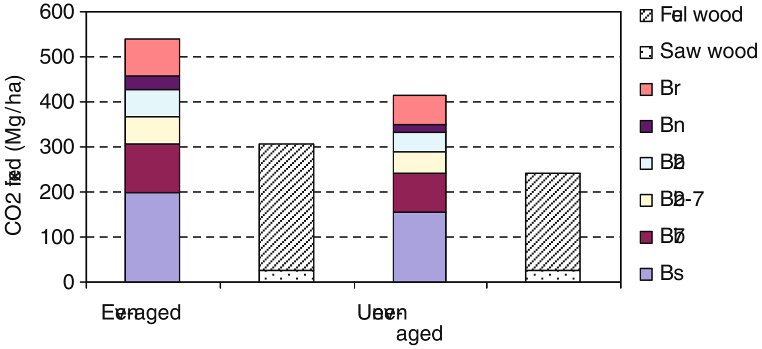


Fig. 3 State and transition scheme of the steppes of the region. From right to left, the changes in structure and composition of the steppe associate with plant cover losses, litter decrease, soil organic matter losses, and erosion

to more irreversible states and ecosystem aridity. For the sub-Andean Patagonia, the states and transitions from rich to poorer range conditions are (Fig. 3):

Grass-steppe → grass-shrub-steppe → shrub-grass-steppe → shrub-steppe

Recently, some biomass and carbon assessments of various types of steppes were made. Laclau (2006), unpublished found values between 14.1 and 22.5 Mg CO_2eq ha^{-1} in shrub-grass steppes of *Festuca pallescens*, *Stipa speciosa*, *Mulinum spinosum* and *Acaena splendens* in the south of Neuquén, consistent with mean values of 24.4 and 25.1 Mg CO_2eq ha^{-1} found by Loguercio et al. (2004) in similar plant communities of Chubut. In tall shrub steppes of *Colletia spinossisima*, *Berberis* spp, *Senecio* spp, *Adesmia volksmanii* and *Baccharis racemosa*, the sequestered carbon scaled up to 56.1 Mg CO_2eq ha^{-1} (Loguercio et al., 2004).

5 Stand Growth, Biomass, and Carbon Estimates

5.1 Ponderosa Pine Growth at Stand Level: Models and Algorithms

The first introduction of ponderosa pine was made by the National Park Administration in the second decade of the 20th century (Tortorelli, 1955; Dimitri, 1972). The seeds came from populations of the west coast of the USA. This origin fitted well with the ecological characteristics of the region, showing the adaptation and a high productive potential of the species (Fig. 4).

Among some early experimental pine plots that still exist an old stand at Isla Victoria, Nahuel Huapi National Park, showed a remarkable growth. When this plantation was 68 years old, it accounted for 1,800 m^3 ha^{-1} volume and 131 m^2 ha^{-1} basal area

Fig. 4 Unpruned, dense ponderosa pine plantation of 40 years age, in Puerto Patriada, Chubut (Photo: F. Letourneau)

Table 1 Structural parameters of Pinus ponderosa plots located in Chubut, Río Negro and Neuquén, Argentina. H (m), dominant tree height; G (m^2 ha^{-1}), basal area; DG (cm), square mean diameter; IS_{20} (m), site index at 20 years abh; N (pl ha^{-1}), standing trees density; V (m^3 ha^{-1}), total stem volume; ABH (year), age at breast height

Location	H	G	DG	SI_{20}	N	V	ABH
40° 03’ S, 71° 04’ W	25	106	41.0	–	800	911	–
40° 55’ S, 71° 33’ W	37	131	44.0	16.0	865	1,862	62
41° 59’ S, 71° 31’ W	21	80	23.9	14.0	1,786	610	33
42° 00’ S, 71° 08’ W	19	107	33.7	12.8	1,200	802	33
42° 09’ S, 71° 31’ W	31	72	36.5	19.5	685	828	36
42° 31’ S, 71° 30’ W	21	103	26.0	15.4	1,957	876	30
43° 07’ S, 71° 34’ W	18	79	23.9	9.8	1,766	625	34
43° 59’ S, 71° 31’ W	35	96	39.0	21.0	819	1,285	40

(Moretti & Fritz, 1989). The number of standing trees *per* hectare was 865, and the dominant tree height 37 m. The same stand was measured eight years after (Laclau, unpublished, 2006), showing a basal area of 160 m^2 ha^{-1}, a dominant tree height of 40 m, and a density of 835 trees per hectare. The site index at 20 years abh was about 16 m. Some other pine plantations, established for scientific or ornamental purpose by the National Forest Administration or by private landowners, also give evidence for the regional performance of the species. Structural parameters of some pine plots assessed along a latitudinal distance of 500 km are shown in Table 1. The SI_{20} estimates correspond to the common range for the region, between 9 and 20 m.

The yield of ponderosa pine stands has been locally assessed by means of permanent or temporary plots studies (Gonda, 1998; Andenmatten & Letourneau, 2003). The latter, adjusted a prediction model based on the relationship between two indexes that relate stand size and plant density: the r*elative density* (Curtis, 1982) and the *space factor* (Hart-Becking *in:* Prodan et al., 1997). The relative density is a coefficient obtained by dividing the basal area (m^2 ha^{-1}) by the square root of the mean quadratic diameter (cm) (Curtis, 1982). The space factor was re-expressed as its inverse, named after *height factor*, and used this way, for modelling simplicity (Andenmatten & Letourneau, 1997). The driving variable of the model is the *dominant tree heigh*, Its dynamics can be predicted through site index curves and the stand age. The estimate of the dominant tree height at any point of time, allows the calculation of the relative density and, with appropriate equations, the deduction of some common descriptive stand variables, like basal area and stand volume. The simplified model (after Mitchell & Cameron, 1985) has the form:

$$V = a \times H^b \times RD^c \tag{1}$$

Where:

V: stand volume (m^3 ha^{-1}),
H: dominant tree height (m),
RD: relative density,
a, b and c: parameters.

The model is used to reflect the response of the stand yield after environment changes affecting some of the involved variables. If the *dominant tree height* or the *relative density* of the stand decrease, the volume yield at any point of time will also decrease, since from Eq. 1 there is a direct relationship between the predictive variables and volume outputs. To anticipate the direction of this response, the relation between the environmental factors and those variables should be known. However, there are no reliable available functions for ponderosa pine in the region, since the SI_{20} values were only loosely connected to any of the considered environmental variables ($R^2 < 50\%$, Andenmatten et al., 2002).

The authors developed the *reference site* method, which compares the values assigned to the variables already assessed by Irisarri and Mendía (1991) – based on Bonfils, 1978 – in unafforested lands, with the same variable scores in neighbouring afforested sites, with a known SI_{20}. Some of the variables account for the availability of soil water, and so, they were used to predict the effect of long-term climate change scenarios that modify the water balance on the stand yield. For example, a rainfall decrease, or the increase of evapotranspiration caused by higher temperatures, will reduce the *available soil water* for plants. This will reduce the score of this variable and subsequently the score for the entire site class. This way it is possible to predict the dominant tree height at a certain age, and through the Eq. 1, to estimate the new expected stand yield. The same applies for the variable *soil moisture at the end of the dry season*. Other considered variables for this method like *aspect, slope* and various soil physical and chemical atributes are more stable environmental properties, so they would not be modified by eventual climate changes, keeping the actual site class score unaffected.

However, although a reduction of the available soil water in a drier climate is plausible, the interpretation of such changes is highly speculative. Climatic features that are not included in the concept also affect the complex interactions of the plant-soil-water system (Easterling et al., 2000). Weather variability can make it difficult to predict climate effects on plant productivity (Fay et al., 2003; Fauchereau et al., 2003), *e.g.*, a change on the annual rain distribution can strongly affect the productivity, especially of low water-storage soils (Ramos, 2006). In the same area as of this study Letourneau et al. (submitted) found for native *Austrocedrus chilensis* that a high annual water recharge, lenghtened the plant growing season and increasing the radial growth. Since this variable is a direct driver of the relative density from Eq. 1, the stand yield is expected to increase.

5.2 Carbon Sequestration in Biomass, Litter and Soils

The model of Andenmatten and Letourneau (2003) provides a regionally reliable estimate for stand yield. To assess the biomass of trees, Laclau (2003) adjusted allometric models based on single tree measures of dense plantation pines of medium quality sites. These equations to calculate the biomass of the different tree compartments follow the form:

$$\log B_i = a + b.\ \log x \tag{2}$$

Where:

B_i: biomass (g) of the i tree compartment, for i = stem, branches, needles, woody roots and tapper
x: explaining variable; conversely *stem volume* (m^3) or *dbh* (cm),
a, b: parameters.

These functions – not regionally validated yet-, were developed for trees of dbh $<$ 35 cm and used to estimate the biomass of some young ponderosa pine stands by means of forest inventories in plots covering a wide variety of topographic and climate situations (Laclau, 2003). Some indicative values for standing biomass of forests about 18–20 years age are shown in Fig. 5. Assuming a carbon content of 50% of the total dry matter (IPCC, 2003), these plantations sequestered about 73 to 330 Mg CO_2eq ha^{-1}, corresponding to a mean capture of about 4 to 18 Mg CO_2eq ha^{-1} $year^{-1}$ depending on the site class.

These biomass estimations positively correlated with the relative density index of a sample of 48 plots. They also fit well in log-linear models ($R^2 > 90$, Laclau, unpublished, 2006), and thus making it feasible to link these estimates with the stand yield model of Andenmatten and Letourneau (2003). That allows to use commonly available parameters of forest inventories for the biomass assessment.

Forests accumulate considerable amounts of litter over time. To the seasonal litterfall, the managed plantations add a significant mass of green foliage and branches after prunings, thinnings and harvest operations. The slow decomposition rates, attributable to the climate and to a high lignine/N ratio of the substrate (Mazzarino et al., 1998; Candan et al., 2003) and the long residence time of the coarse woody debris

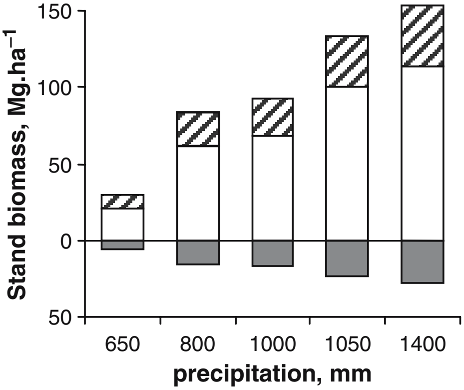


Fig. 5 Tree biomass of ponderosa pine stands, estimated from single tree models (Laclau, 2003) applied to forest inventories, under different rainfall condition. The hatched sectors represent the crown biomass, the empty sectors, the stem biomass, and the grey ones, the root biomass (Laclau, 2006 unpublished)

lead to high carbon stocks in the forest floor. For example, the pool of fine litter (dry needles) found in plantations of the same range as the ones shown in Fig. 5, was 11.0 ± 2.32 Mg ha^{-1} (mean ± se), and the pool of woody debris was 6.9 ± 1.65, both representing 32.7 ± 7.32 Mg CO_2eq ha^{-1} of carbon (Laclau, unpublished, 2006). For soils, there is not yet enough evidence regarding a different carbon capture of the forest plantations as compared to the replaced steppe (Buduba et al., 2002; Laclau, 2003; Candan et al., 2003). No differences were detected between thinned and unthinned paired plots of pine plantations of about 20 years, after two years of this intervention (Laclau, unpublished, 2006). In a regional assessment, the carbon stored in a 50 cm profile of rather allophanic soils, fluctuated between 316 ± 24 Mg CO_2eq ha^{-1} for ponderosa pine plots, and 340 ± 200 Mg CO_2eq ha^{-1} for grass-shrub steppes (Laclau, 2003). However, these large amounts of soil carbon – closely related to the parent material and the hydrological regime – could shade a minor carbon content in the upper organic horizon of pine plantations with respect to grass-shrub steppes soils, observed in preliminary assays (Candan et al., 2003).

6 Regional Scenarios of Carbon Sequestration

6.1 Carbon Capture at the Stand Level in Managed Forests

The contribution of local forest plantations to climate change mitigation can be quantified as a product between two terms: the afforested area and the amount of carbon sequestered over time over a local baseline, in a *per* unit area basis. The last term can be assessed through the model of Andenmatten and Letourneau (2003), that simulates the stem and stand density dynamics along the rotation, allowing the calculation of

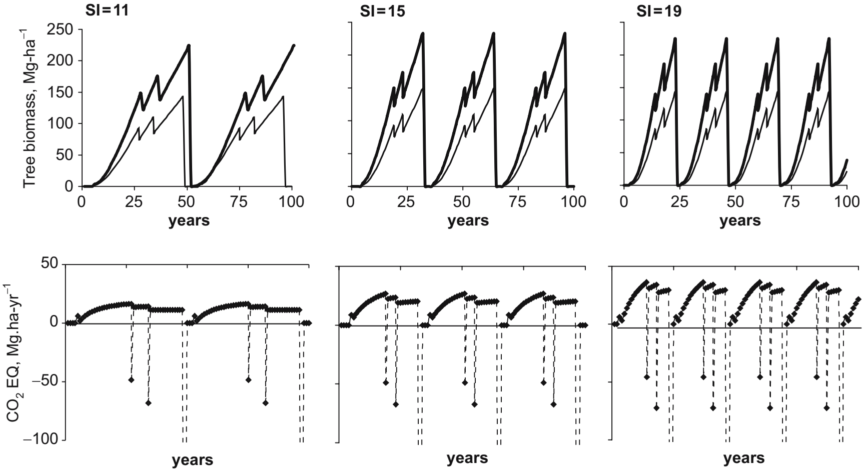


Fig. 6 Managed ponderosa pine rotations in three site classes (above), and their correspondent CO_2eq accumulation rates (below). At a low site quality (SI_{20} = 11) the rotation lasts 48 years, in the intermediate (SI_{20} = 15) 32 years, and in the best site (SI_{20} = 19), 23 years. In the upper graphs, the thick line represents the total stand biomass, and the fine line, the stem biomass only; the difference between both curves is the crown and roots biomass. In the bottom graphs, the abrupt fall of the rates at the time of thinning and harvest is shown

biomass or carbon stocks by means of allometric equations. Figure 6 is an output of this application, and shows the accumulated biomass of successive rotations managed under similar management regimes in three site classes, based on simulations of volume growth (Laclau et al., 2003) in a 100-year projection. The stand growth after plant establishment involves a continuous carbon accumulation in the forest biomass, eventually disrupted by interventions that partially (pruning and thinnings) or totally (harvest) reduce the carbon stocking, and recurring in the next rotations. The growing rates rise as the site quality increases (Fig. 6), shortening the production cycle, allowing for more rotations in the same time interval, and ultimately leading to a higher rate of CO_2 sequestration. The *mean* sequestered carbon along a 100-year period, resulted in 256, 278 and 286 Mg CO_2eq ha^{-1} in sites SI_{20} = 11, SI_{20} = 15 and SI_{20} = 19, respectively. Besides that slightly higher accumulation in the more suitable sites, the mean sequestered carbon of the period was reached earlier (after: 19 years in SI_{20} = 19, 25 years in SI_{20} = 15, and 33 years in SI_{20} = 11).

6.2 *Economic Aspects of Carbon Sequestration at the Watershed Level*

Loguercio et al. (2004) analysed the afforestation potential of a southern area for producing timber to supply at-scale industries of a forest cluster and also benefit carbon capture. The forest plantations, still young in most of the region, would be established at sites that are economically marginal for cattle breeding, in the vicinity of urban settlements, which would provide a ecological services and concentrate the industry demand.

The study area is a part of the forest-steppe ecotone of the northwestern Chubut province and comprises about 154,000 ha of land around the town of Cholila (Fig. 1). It includes native forests and woodlands (52,000 ha), grass meadows (10,000 ha), grass and shrub-steppes (68,000 ha) and other bare lands or open water (24,000 ha) (Fig. 7). The dominant soils are allophanic or other volcanic types, with materials deposited over, or mixed with, glacial till (Veblen & Lorenz, 1988). The vegetation gradient is pronounced, the same as in the rest of the region, changing from *Nothofagus* and *Austrocedrus chilensis* forests to grass-shrub steppes in a distance of no more than 50 km, while the rainfall consistently decreases from the Andean cordillera to the arid Patagonian steppe (from near 2,000 mm year^{-1} to 400 mm year^{-1}, Farias, 2003).

The town of Cholila has almost 2,000 inhabitants (INDEC, 2001), and in the sorroundings there are about 62 rural settlements. A few farms *(estancias)* to the east own between 5,000 and 50,000 ha each. To the west of town the farms have less than 3,000 ha (Loguercio et al., 2006). Even though from the late last century there are evidences of land concentration 44% of the properties have less than 500 ha.

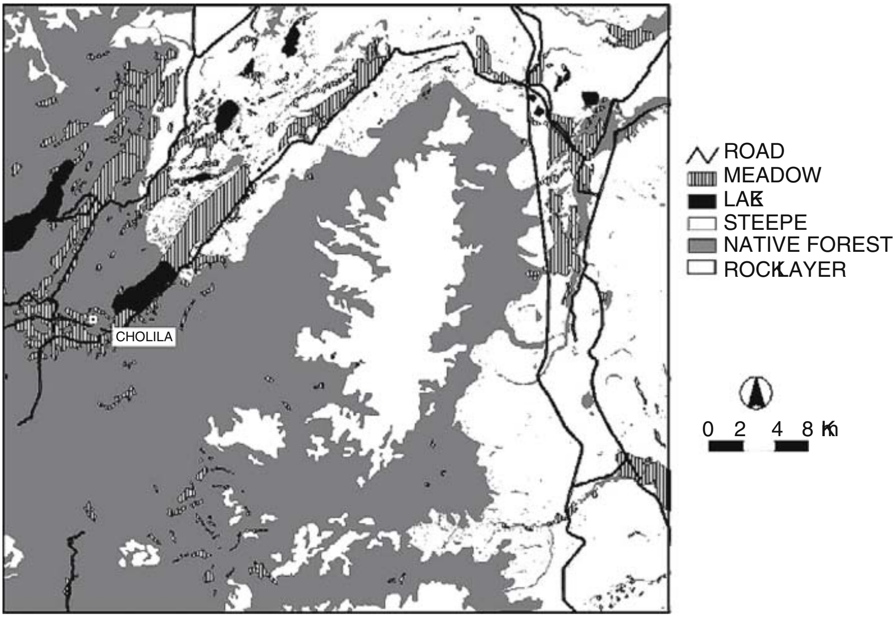


Fig. 7 Map of the environmental units of Cholila area. Steppe lands (light gray units) are considered afforestable for ponderosa pine

The main economic activity is the extensive cattle and sheep ranching although it is strongly affected by the harsh environment (Peralta, 1995). To the east, major sheep farms dominate in lands of low carrying capacity; instead, in the western more fertile lands, cattle breeding impose. The patches of grass meadows, which produces four to ten times more forage than the contiguous steppe (Martínez Crovetto, 1980), strongly improves the carrying capacity of the land. The bovine stock is usually lower than 200 heads *per* farm, and the sheep stock – historically decreasing – is lower than 1,500 heads, except for the big farms (Danklmeier, unpublished, 2006). The land carrying capacity is highly variable between zones and farms, and modal estimates are in the range of 0.1–0.2 bovine units *per* hectare, or 0.25–0.50 sheep *per* hectare (Guitart, et al., unpublished, 2007).

At present, there is an increasing touristic development in the region. The so-called "Patagonia brand" promoted new projects in the sector, some cases related to rural activity. On the other hand, afforestation is still young, accounting for only a few young pine plantations. In spite of this, several landowners claim for forest development – which is promoted by different kind of public policies – as a diversification strategy, what brings the motivation for the following analysis.

To assess the potential of the area, a scenario based on the afforestation on suitable land free of native forests since a long time was adopted. The suitable land for afforestation were classified using a predicting function of the *internode index* – a site-quality indicator, defined as the length of five internodes over the dbh of the averaged dominant trees –, developed for the afforestable area of Chubut province (Loguercio et al., 2004). The predicting variables were the *mean annual rainfall*, the site *aspect* and *slope*, spatially represented by digitalized isohiets and a digital elevation model. The size of each site class is given in Table 2.

The potential gain of carbon was calculated as the difference between the baseline and the sequestered carbon by plantations. The forest growth was simulated with the algorithm of Andenmatten and Letourneau (2003, *see subchapter 4.1*). The assumed timber production management includes prunings and thinnings interventions. The stand density index of Reineke was maintained between 700 and 500 (Gonda, 2001). For classes I and II, the target dbh for clearfelling harvest was 50, and 40 cm for III and IV classes. The above- and belowground biomass was calculated with allometric functions locally adjusted for ponderosa pine (Loguercio, et al., unpublished, 2002).

Table 2 Site class, internode index (II, m), SI_{20} (m), rotation period (year) and land extension (ha, %) for ponderosa pine at the study region

				Land extension	
Site class	II	SI_{20}	Rotation period	ha	%
I	4.3	17.1	38	2,025	3.0
II	3.5	14.8	47	20,015	29.3
III	2.7	12.4	49	26,740	39.1
IV	1.9	10.0	62	19,548	28.6
Total	2.8[a]	12.6[a]	52[a]	68,328	100.0

[a] Mean weighed values for the whole afforestable area

The baseline is represented by the carbon dynamics of the existing steppe vegetation. To assess the steppe environment, a physiognomic type cover (*sensu* Anchorena, *in:* Beeskow et al., 1987) was applied to *Aster* image analysis. This analysis consisted of an un-supervised classification based on a binary-hierarchical method, with field checking (Loguercio et al., 2004). The dominant vegetation types were the grass steppes of *Stipa speciosa var. major* and *Festuca pallescens* (36% of the total steppes) and the shrub steppe of *Mulinum spinosum, Senecio spp. and Berberis spp*. (25%). The minor types were the thickets of *Rosa eglanteria* shrubs (3%), and the dwarf-shrub steppe of *Nassauvia glomerulosa, Acaena splendens and Azorella monanthos* (3%). The baseline was conservatively assumed as the above- and belowground carbon increase that could occur in the existing steppe along a period similar to that of the rotation period of the plantation, weighed by the relative extension of each steppe type. This increment – representative of each plant community –, was estimated as the absolute value of the difference between the maximum (mean plus standard deviation) and minimum (mean minus standard deviation) of the observed distribution values (Loguercio et al., 2004, Table 3).

The analysis scenario was based on the following assumptions:

- Thirty percent of the land extension free of native forests or woodlands was planted with ponderosa pine (20,500 ha), and the proportion for each site class was the same as for the total afforestable lands of the area (Table 2).
- The afforestation gradually extended along ten years, at a rate of 2,050 ha *per* year, with continuing reforestation after each rotation.
- There were no carbon leakages due to any effect of the forestry, except for those caused by forest management.

The output showed that the afforested area will stabilize around 20,000 ha, with minor fluctuations after the first harvests. The carbon stock will grow along the first rotation to reach about 6.7 Gt CO_2eq, and then oscillate in the succesive turns of harvest and planting, between 3 and 6 Gt CO_2eq (Fig. 8). In spite of the assumed increase, the baseline will remain low (Fig. 8). The net carbon capture (afforestation

Table 3 Accumulated carbon per hectare (CO_2eq Mg ha^{-1}, mean and standard deviation), expected increase of the baseline (CO_2eq Mg ha^{-1}), and land extension (ha and %) for each vegetation type

	Accumulated carbon			Land area	
Vegetation type	Mean	SD	CO_2eq incr	(Ha)	(%)
Shrubs (several spp.)	56.1	12.9	25.9	16,557	24.2
Rosa eglanteria thickets	34.8	13.6	27.1	2,056	3.0
Shrub steppe	25.3	10.1	20.1	16,964	24.8
Dwarf-shrub steppe	25.3	10.1	20.1	1,759	2.6
Grass steppe	24.6	9.9	19.8	24,546	35.9
Degraded steppes	13.0	5.6	11.2	6,446	9.4
Total	31.6[a]	10.4[a]	20.7[a]	68,328	100.0

[a] Mean weighed values for the whole afforestable area

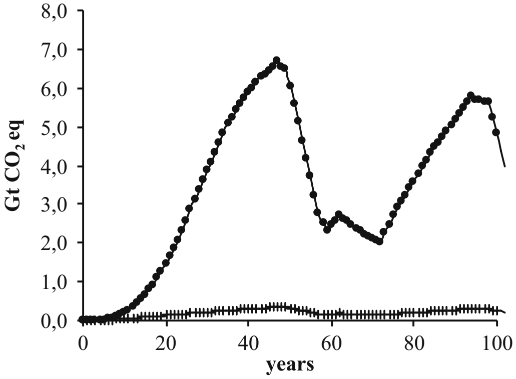


Fig. 8 Carbon sequestering scenario for 20,500 ha afforested lands with ponderosa pine during succesive rotations (black-circled line) and baseline (crossed line)

gain minus baseline), after a peak of 6.3 Gt $MgCO_2$eq at 50 years, will vary between 2 and 6 Gt CO_2eq.

6.3 Carbon Capture at Regional Level

In a spatial analysis, *per* hectare values of CO_2 sequestered would be additive if the land would be afforested at the same time and under the same management regime. This would implicate 100% of CO_2 emissions at the harvest time, if carbon stored in long term wood products were not considered. However, a sound strategy for climate change mitigation should aim at maximizing the sink effect, *i.e.*, to take up more atmospheric carbon than under the actual land use, and once this goal was achieved, keep the carbon sequestered in the long run. Forest planning based on sustainable yields is consistent with this strategy and also prevents timber supply fluctuations. To meet these objectives the plantations layout would be gradual and sustained, creating a forest landbase such as, once the equilibrium is reached, flows of forest goods and services remains constant.

Assuming a sustained plantation of land patches proportional to the total available land, the sink effect at a regional level would be maximum, the same as the CO_2 conservation. *E.g.*, if the plantation turn is 40 years, and the available forest land is 100,000 ha, a linear plantation rate of 2,500 ha year^{-1} will cover the whole area at the harvest time of the first plantations, stabilizing then the mean growth of the all set planted alongtime. *I.e.*, in the first year of a project, only 2.5% of the total area will be planted, 5% at the second, and so on, until 100% land occupation are reached after 40 years. At that point 2.5% of the plantation area will be harvested for the first time. Based on this assumption, the temporal rotation sequence should be as indicated in Fig. 9a. The *accumulated mean* biomass for each site should be as indicated in Fig. 9b.

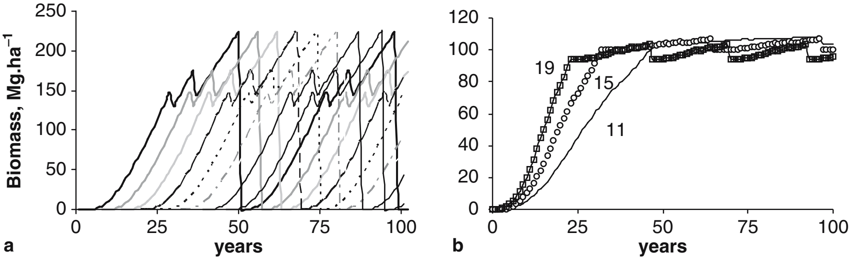


Fig. 9 (a) Stand biomass curves of succesive rotations of ponderosa pine, and (b) accumulated mean biomass curves for three site qualities: $SI_{20} = 19$ (squared line), $SI_{20} = 15$ (empty circles) and $SI_{20} = 11$ (line)

On this basis, three alternative regional scenarios of CO_2 capture were analysed: a conservative forest development (E1), keeping the present plantation rates for the region (Loguercio & Deccechis, 2006) of about 5,000 ha year^{-1}. At the end of one rotation period, forests would cover 6% of the potential forest land; an intermediate scenario (E2) with a constant plantation rate of 13,000 ha year^{-1} and a final forest cover of 15% of the area, and an optimistic scenario (E3), assuming a plantation rate of 26,000 ha year^{-1} and a final 30% occupation of the afforestable land. The assumptions of the simulation are:

- All rotations were managed under the same prescriptions of prunings and thinnings, and clearfelling. The opportunity and intensity of each intervention is defined by maximum and minimum thresholds of relative density, and dbh targets. These thresholds are reached at different points in time, depending on each SI_{20} (Laclau et al., 2003).
- The extent and suitability of the afforestable land was based on the above mentioned studies of site quality *(see subheader 3)*, alternatively assigning to them the site indexes: $SI_{20} = 19$ *(very suitable* and *suitable* land), 15 *(moderately suitable)* and 11 *(poorly* and *marginally suitable).*
- The increase of the planted area was linear for each scenario, and the plantations allocated proportionally to the existing site classes.
- The annual losses of plantations were randomly assumed as a proportion from 5% to 50% for afforested lands of $SI_{20} = 11$; from 5% to 40% for lands of $SI_{20} = 15$, and from 5% to 30% for lands of $SI_{20} = 19$. These values gave account of forest baseland reductions or stock losses due to environmental contingencies – including negative effects of the climate change – or unappropriate management.

At steady state, the simulation output showed an average sequestered carbon level of 24.4 Gt CO_2eq for scenario E1; 61.7 for E2, and 120.1 for the scenario E3 (Fig. 10a). The total afforested land – after a mean weighed rotation period of 34 years – reached up to 170,000, 425,000 and 850,000 ha, for scenarios E1, E2 and E3, respectively (Fig. 10b).

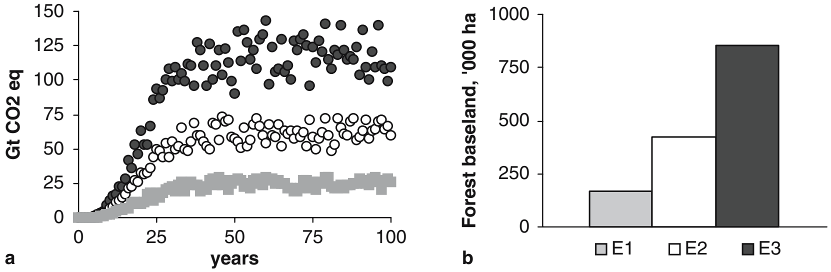


Fig. 10 Simulation output for regional scenarios E1 (gray symbols), E2 (empty black-outlined symbols) and E3 (black filled symbols); (a) carbon sequestration, (b) forest baseland

7 Conclusions

The regionally valid models for stand yield and for single tree biomass of ponderosa pine allow the assessment of the possibilities of the region to contribute to climate change mitigation through forestry, in sites of low carbon level at the present time. By means of recently developed models, it is feasible to simulate forest scenarios at multiple scales, from individuals or stands to forest landscapes. Also, the *site reference* method could give account of the shift of the site quality after climate changes affecting some edaphic variables related to forest productivity.

The proposed scenario for Cholila shows how the forest cover of the region could develop in a sustained way. Since a considerable extension of grasslands have several constraints to breed cattle due to overgrazing, reduced forage yield or lack of drinking water sources (Paruelo et al., 1992; Golluscio et al., 1998), a reasonable scenario based on 30% of steppe replacement by forest plantation could accomplish for both beef and pine production. The need to prevent grazing from the afforested sites requires paddocks fencing, a major feature for a sustainable cattle management of patagonian steppes (Golluscio et al., 1998). The eventual displacement of cattle from sites devoted to plantations would not neccesarily compete with beef production since no direct competition exists between site qualities for forestry and for cattle (Laclau, unpublished, 2006). On the other hand, forestry diversification could increase the income of landowners and promote the development of the local industry. The exclusion of sites with native forests or potentially recoverable for natural regeneration would conserve the carbon stored in such native ecosystems and eventually help to reduce or re-allocate the actual animal stocking, subsequently augmenting the land use efficiency. To minimize carbon leakages and prevent some negative environmental impacts, the layout of plantations should keep the functionality and integrity of native ecosystems (Rusch & Schlichter, 2005; Rusch et al., 2005a). Besides, some positive impacts like soil erosion prevention, or the creation of appropriate conditions for tree diversification by planting under forest cover could be reached.

At present, the afforestation with ponderosa pine and other conifers in Patagonia is made at minimum rates, very below its agroecologic and economic potential. The creation of forest production clusters would strongly contribute to atmospheric carbon capture. Also, the economic development would ease the increase of plantations in sites of marginal condition, which are now admissible under the Clean Development Mechanism of the Kyoto Protocol to credit carbon emissions reductions.

Acknowledgements The studies supporting this chapter have been carried out by the Instituto Nacional de Tecnología Agropecuaria (INTA) and the Centro de Investigación y Extensión Forestal Andino-Patagónico (CIEFAP) with funding help of the German Technical Cooperation Agency (GTZ) and the Forest Development Project of the Secretaría de Agricultura, Ganadería, Pesca y Alimentación (SAGPyA). J. Lencinas, S. Havrylenko and S. Antequera from CIEFAP, helped to assess the Cholila study case. R. Jandl made valuable suggestions to improve the manuscript.

References

Andenmatten, E., and F.J. Letourneau (1997). Curvas de Indice de Sitio para Pinus ponderosa (Dougl.) Laws, de aplicación en la región Andino Patagónica de Chubut and Río Negro, Argentina. *Bosque 18*(2): 13–18.

Andenmatten, E., and F.J. Letourneau (2003). Predicción y proyección del rendimiento de pino ponderosa en las provincias de Chubut and Río Negro, Argentina. *Revista Quebracho*. Universidad Nacional de Santiago del Estero, Santiago Del Estero, Argentina vol. 10, 23 p.

Andenmatten, A., C. Lopez, and F.J. Letourneau (2002). Método del sitio de referencia, para la estimación predial de calidad de sitio. *Comunicación Técnica* No. 21. INTA, EEA Bariloche, Área Forestal, Silvicultura, 10 pp.

Armesto, J.J., C.VIllagran, and M.K. Arroyo, Editors (1997). *Ecología de los Bosques Nativos de Chile*. Editorial Universitaria, Santiago (Ch), 470 pp.

Assmann, E. (1970). *The principles of forests yield study*. Pergamon Press, New York, 506 pp.

Austin, A., and O. Sala (2002). Carbon and nitrogen dynamics across a natural precipitation gradient in Patagonia, Argentina. *Journal of Vegetation Science 13*: 351–360.

Beeskow, A.M., H. del Valle, and C.M. Rostagno (1987). *Los sistemas fisiográficos de la región árida y semiárida de la Provincia del Chubut*. Secretaría de Ciencia y Técnica, Delegación Regional Patagonia, Bariloche, Argentina.

Bertiller, M.B., and A. Bisigato (1998). Vegetation dynamics under grazing disturbance. The state-and-transition model for the Patagonian steppes. *Ecología Austral* 8: 191–199.

Bonfils, P. (1978). Le classement des sols en vue de la reforestation en zone mediterranéenne. *Rev. Forest. Fr. 30*(4): 271–282.

Broquen, P., Girardin J.L., and G. Falbo (1998). Soil features related with Pinus ponderosa, Dougl. growth. *Symposium of Sustainability of soil fertility under forests and indicators to measure soil changes*. 16th World Congress of Soil Science, 5 pp.

Buduba, C.G., G.A. Loguercio, J.A. Irisarri, T. Voigt, and M.F.Valenzuela (2002). Evaluación preliminar del contenido de carbono edáfico en plantaciones de pino ponderosa en el oeste del Chubut. *En Actas de XVIII Congreso Argentino de la Ciencia del Suelo*, 6 pp.

Candan, F., P. Broquen, and G. Falbo (2003). Chemical soil properties in Humic Vitrixerand and Vitrandic Haploxerol under ponderosa pine and steppe (SW Neuquén). Revista de Investigaciones Agropecuarias INTA, Argentina *32*(1): 71–88.

Colmet-Dâage, F. (1992). Principales áreas forestales de Argentina, productividad elevada y sostenida. *VIII Simposio de la Sociedad Chilena de la Ciencia del Suelo*, Valdivia, Chile, 88 pp.

Cordon, V., J.C. Forquera, J. Gastiazoro, V. Barros, and I. Schmidt (1993). *Estudio microclimático del área cordillerana del Sudoeste de la Provincia. de Río Negro. Cartas de precipitación*. Facultad de Ciencias Agrarias, Cinco Saltos, Río Negro.

Corley, J., P. Sackmann, V. Rusch, J. Bettinelli, and J. Paritsis (2005). Effects of pine silviculture on the ant assemblages (Hymenoptera: Formicidae) of the patagonian steppe. *Forest Ecology and Management 222*: 162–166.

Curtis, R.O. (1982). A simple stand index of stand density for Douglas-fir. *Forest Science 27*(1): 92–94.

Davel, M., and A. Ortega (2003). Estimación del índice de sitio para pino oregón a partir de variables ambientales en la Patagonia Andina Argentina. *Bosque 24*(1): 55–69.

Dimitri, M.J. (1972). La Región de los Bosques Andino-Patagónicos. Sinopsis General. *INTA, Colección Científica*, T. X, Buenos Aires, 381 pp.

Dinerstein, E., D.M. Olson, D.J.Graham, A.L.Webster, S.A. Primm, M.P. Bookbinder, and G. Ledec (1995). *Una Evaluación del Estado de Conservación de las Ecoregiones Terrestres de América Latina y el Caribe*. WWF/Banco Mundial, Washington, DC, pp. 135.

Easterling, D.R., J.L. Evans, P. Groisman, T.R. Karl, K.E. Kunkel, and P. Ambenje (2000). Observed variability and trends in extreme climate events: A brief review. In: 'Understanding Changes in Weather and Climate Extreme' Series. *Bulletin of the American Meteorological Society 81*(3): 417–425.

Farias, F. (2003). *Land cover change in a rural area of the forest-steppe ecotone of Andean Patagonia, Argentina: Utilizing Landsat data for the detection and analyzing the change*. M.Sc. thesis, Lund University, 50 pp.

Fauchereau, N., S. Trzaska, M. RouaulT, and Y. Richard (2003). Rainfall variability and changes in southern Africa during the 20th century in the global warming context. *Natural Hazards 29*(2): 139–154.

Fay, P.A., J.D. Carlisle, A.K. Knapp, J.M. Blair, and S.L. Collins (2003). Productivity responses to altered rainfall patterns in a C4-dominated grassland. *Oecologia 137*(2): 245–251.

Ferrer, J.A., J.A. IrisarrI, and J.M. Mendía (1990). *Estudio regional de suelos de la provincia de Neuquén*. Inédito. COPADE/ Consejo Federal de Inversiones, Argentina, vol. 5, Tomo I, 244 pp.

Gyenge, J.E., M.E. Fernández, G. Dalla Salda, and T.M. Schlichter (2002). Silvopastoral systems in NW Patagonia: II. Water balance and water relations in a stand of Pinus ponderosa and native grassland. *Agroforestry Systems 55*: 47–55.

Gyenge, J.E., M.E. Fernández, and T.M. Schlichter (2003). Water relations of ponderosa pines in Patagonia Argentina: implications on local water resources and individual growth. Trees: structure and Function 17(5): 417–423.

Golluscio, R.A., V.A. Deregibus, and J.M. Paruelo (1998). Sustainability and range management in the Patagonian steppes. *Ecología Austral 8*: 265–284.

Gonda, H. (2001). Modelo preliminar de manejo para plantaciones de pino ponderosa en sitios de calidad media en la Patagonia andina. *Revista Patagonia Forestal*, CIEFAP, año VII(3): 7–10. http://www.ciefap.org.ar.

Gonda, H.E. (1998). *Heigth-diameter and volume equations, growth intercept and needle length site quality indicators, and yield equations for young Ponderosa pine plantations in Neuquén, Patagonia, Argentina*. Ph.D. thesis, Oregon State University, OR, 198 pp.

Instituto Nacional de Estadística y Censos (INDEC) (2001). *Censo Nacional de Población, Hogares y Viviendas del año 2001*. http://www.indec.gov.ar/webcenso/index.asp.

Intergovernmental Panel on Climate Change (IPCC) (2003). *Good practice guidance for land use, land-use change and forestry*. Institute for Global Environmental Strategy (IGES). http://www.ipcc-nggip.ipes.or.jp

Irisarri, J.A., and J.M. Mendía (1991). *Reconocimiento de Suelos y Evaluación de la Aptitud Forestal de la Región Precordillerana de la Provincia de Río Negro*. Consejo

Federal de Inversiones/ Facultad de Ciencias Agrarias de la Universidad Nacional del Comahue, pp. 61.

Irisarri, J.A., J.M. Mendia, C. Roca, C. Bucuba, F. Valenzuela, F. Epele, F. Fraseto, G. Ostertag, and E. Andenmatten (1997). *Zonificación de las tierras para la forestación. Provincia del Chubut*. Inédito. Consejo Federal de Inversiones, Argentina, 150 pp.

Kitzberger, T., T.T. Veblen, and R. Villalba (1997). Climatic influences on fire regimes along a rainforest-to-xeric woodland gradient in northern Patagonia, Argentina. *Journal of Biogeography 24*: 35–47.

Laclau, P. (2003). Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in northwest Patagonia. *Forest Ecology and Management 180*: 317–333.

Laclau, P., L.M. Pozo, G. Huerta, E. Andenmatten, and F. Letourneau (2003). Rentabilidad de la forestación con pino ponderosa (Pinus ponderosa (Dougl.) Laws) en el noroeste de la Patagonia, Argentina. *Bosque 23*: 21–36.

Loguercio, G.A., and F. Deccechis (2006). Forestaciones en la Patagonia andina: potencial y desarrollo alcanzado. *Revista Patagonia Forestal*, CIEFAP, marzo 2006: 4–6. http://www.ciefap.org.ar.

Loguercio, G.A., J.D. Lencinas, S. Antequera, G. Defossé, M. Gimenez, H. Gonda, D. Martinez, F. Ríos Campano, and A. Siebert (2004). *Bases estratégicas para proyectos de forestación en la Patagonia como Mecanismo para un Desarrollo Limpio. Estudio de caso en la provincia del Chubut*. Informe Final. PIA 05/02, PFD/SAGPyA-CIEFAP, Esquel, Chubut, 72 pp. and anexes.

Loguercio, G.A., G. Salvador, and S. Antequera (2005). Estimación preliminar de producción y rentabilidad de certificados de reducción de emisiones temporales en forestaciones de pino ponderosa en la Patagonia. *En Actas 3° Congreso Forestal Argentino y Latinoamericano*. ISSN 1669–6786, 11 pp.

Loguercio, G.A., M.V. Fernandez, M. Molina-Tagle, M. Davel, and S. Havrylenko (2006). *Modelo de asignación de recursos para la generación de cuencas de forestación en la provincia del Chubut*. Proyecto Forestal de Desarrollo PIA /04. SAGPyA-CIEFAP, 129 pp.

Martinez Crovetto, R. (1980). *Apuntes sobre la vegetación de los alrededores del Lago Cholila (Noroeste de la Provincia del Chubut)*. Publicación Técnica número 1. Facultad de Ciencias Agrarias. Universidad Nacional del Nordeste, Corrientes, República Argentina.

Mazzarino, M.J., M. Bertiller, T. Schlichter, and M. Gobbi (1998). Nutrient cycling in Patagonian ecosystems. *Ecología Asutral 8*: 167–181.

Mendía, J.M., and J. Irisarri (1986). *Relevamiento de suelos con aptitud forestal en la región occidental de la provincia de Río Negro*. Inédito. Consejo Federal de Inversiones, Argentina, 73 pp. and 4 anexes of 36 pp.

Mitchell, K.J., and I.R. Cameron (1985). *Managed stand yield tables for coastal Douglas-Fir: Initial density and precommercial thinning*. Land Management Report Number 31. Ministry of Forests (B.C.), Research Branch, 69 pp.

Moretti, A., and G. Fritz (1989). *Estudio dasométrico de una parcela experimental de Pinus ponderosa, en Isla Victoria, Parque Nacional Nahuel Huapi*. Inédito. Informe interno, Empresa Forestal Rionegrina.

Oesterheld, M., M.R. Aguiar, and J.M. Paruelo (1998). Ecosistemas patagónicos. *Ecología Austral 8*: 75–84.

Paruelo, J.M., R.A. Golluscio, and V.A. Deregibus (1992). Manejo del pastoreo sobre bases ecológicas en la Patagonia extra-andina: una experiencia a escala de establecimiento. *Anales de la Sociedad Rural Argentina 126*: 68–80.

Paruelo, J.M., M.B. Bertiller, T.M. Schlichter, and F. Coronato (1993). *Secuencias de deterioro en distintos ambientes patagónicos. Su caracterización mediante el modelo de estados and transiciones*. Convenio Argentino-Alemán de Cooperación Técnica INTA-GTZ. Proyecto LUDEPA-SME, San Carlos de Bariloche, 110 pp.

Paruelo, J.M., A. Beltran, E. Jobbagy, O. Sala, and R.A. Golluscio (1998). The climate of Patagonia: General patterns and controls on biotic processes. *Ecología Austral 8*: 85–101.

Peralta, C. (1995). *Evaluación del factor antropógeno en el proceso de la desertificación en la Patagonia. Lucha contra la desertificación en la Patagonia*. Inédito. Proyecto de Cooperación Técnica Argentino-Alemana (INTA-GTZ).

Prodan, M., R. Peters, F. Cox, and P. Real (1997). *Mensura forestal*. IICA, 586 pp.

Ramos, M.C. (2006). Soil water content and yield variability in vineyards of Mediterranean northeastern Spain affected by mechanization and climate variability. *Hidrological Process 20*(11): 2271–2283.

Rothkugel, M. (1916). *Los Bosques Patagónicos*. Oficina de Bosques y Yerbales, Ministerio de Agricultura, Buenos Aires, 207 pp.

Rusch, V., and T. Schlichter (2005). *Forest management practices adjusted to conservation goals in Patagonia, Argentina*. Abstracts of IUFRO Congress, Brisbane, Australia, August 2005.

Rusch, V., T. Schlichter, and J. Corley (2005a). *Integration of spacial scales for the conservation of biodiversity in forest plantations*. Abstracts of the International Conference: Biodiversity and Conservation Biology in Plantation Forests. Bordeaux, France, 26–29 April 2005, pp. 66.

Rusch, V., M. Sarasola, and C. Peyrou (2005b). La biodiversidad en bosques de ciprés y ante su reemplazo por pinares. Resumen. In: *Actas de la I Reunión sobre ecología, conservación y uso de los bosques de ciprés de la cordillera*. Esquel, Chubut, 28 al 30 de abril de 2005.

Secretaría de Ambiente y Desarrollo Sustentable (SAyDS) (2005). *Primer Inventario Nacional de Bosques Nativos*. Informe Regional Bosque Andino Patagónico, Proyecto de Bosques Nativos y Áreas Protegidas, BIRF 4085-AR, 204 pp.

Shoji, S., M. Nanzyo, and R.A. Dahlgren (1993). *Volcanic ash soils. Genesis,* properties and utilization. Developments in soil science 21. Elsevier, Amsterdam, The Netherlands, 287 pp.

Tortorelli, L.A. (1955). Lo biológico y lo económico en parques nacionales. *Natura, Administración General de Parques Nacionales* T.*1*(2): 235–245.

Veblen, T.T., and D.C. Lorenz (1988). Recent vegetation changes along the forest/steppe ecotone of northern Patagonia. *Annals of the Association of American Geographers 78*: 93–111.

Veblen, T.T., T. Kitzberger, B.R. Burns, and A. Rebertus (1997). Perturbaciones y dinámica de regeneración en bosques andinos del sur de Chile and Argentina. In: Armesto, J.J., C. Villagrán, and M.K. Arroyo, Editors. *Ecología de los Bosques Nativos de Chile*. Editorial Universitaria, Santiago, pp. 169–197.

Assessing Pine Wilt Disease Risk Under a Climate Change Scenario in Northwestern Spain

G. Pérez[1], J. J. Díez[2], F. Ibeas[2], and J. A. Pajares[2]*

1 Introduction

Forest ecosystems are characterized by their structural complexity and biodiversity and trophic relationships within them commonly involve several levels. Functioning of these systems is likely to be perturbed in many ways if significant warming predicted eventually occurs. Among the interactions to be shifted by temperature increments in the Mediterranean region, these between forest pest and host trees are highly relevant to forest conservation and management, since perturbations may result in many cases in a reduced probability of tree or stand survival.

Expected effects due to warming of the insect-plant interactions in South Europe forests would apply to pole ward and upward shifts in species ranges, particularly of those pests with distributions currently limited by low temperature thresholds, as for example in the pine processionary moth, *Thaumetopoea pityocampa*, a defoliator able to thrive and develop during the winter by behavioural adaptations such as group strategy and the formation of a heat accumulating silk tent. These adaptations allow this species to be installed in most temperate pine forest areas of southern Europe, though caterpillars will not survive inside their tents if air temperature downs below −12°C, excluding processionary moth populations from many mountain areas. Rising of the lower threshold by temperature increase would result in many pine stands now available to be attacked and damaged by *T. pityocampa* populations (Battisti et al., 2005).

Prognosis of climate change in Southern Europe point to a worsening of conditions, with significant higher temperatures and drought, thus a reduction of water availability or an increase of xericity is predicted (IPCC, 2007). Forest trees not particularly adapted to these new conditions would experience water stresses resulting in a reduction of vigour, which in turn will increase susceptibility to lethal pests such as the stem borers. Several semi aggressive bark beetles species as *Ips* or *Tomicus* spp.

---

[1] Researcher at Centro de Sanidad Forestal de Calabazanos, Junta de Castilla y León, Spain

[2] Professors at Dpto. Producción Vegetal y Recursos Forestales, ETS Ingenierías Agrarias, University of Valladolid, Palencia, Spain, *jpajares@pvs.uva.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

are normally restrained to colonize weakened, dying or recently dead trees, but aggressive attacks to living healthy trees will arise if eruptive thresholds are surpassed. A higher beetle density is required for a mass attack if tree vigour is high, thus increment of xeric conditions would surely lead to sustained stress conditions for many pine stands, and to the lowering of the eruptive thresholds, likely promoting a higher frequency of destructive outbreaks by bark beetles (Williams & Liebhold, 2002)

Besides the above mentioned effects, more complex, three level interactions could be altered by warming, as for instance in the insect vector-pathogen-plant interactions. Here we will assess the risk of development of pine wilt disease, a destructive disease caused by a pathogenic nematode and transmitted by a longhorn beetle, and will examine the expected changes in risk rating under a climate change scenario.

The pine wood nematode (PWN), *Bursaphelenchus xylophilus* (Steiner & Buhrer) Nickle, is the causal agent of pine wilt disease, a serious threat that has caused extensive mortality of native pines in Japan and East Asia since 1900s (Mamiya, 1988). It is vectored to new host trees by cerambycid beetles in the genus *Monochamus* (pine sawyer beetles) and healthy trees are inoculated during adult maturation feeding in the pine shoots by the sawyers, as the nematodes emerge from the insect vector and enter the trees through the feeding wounds. Infected trees die by the action of the nematodes and oviposition by *Monochamus* females later occurs on these dead trees, larvae burrowing under the bark and entering into the wood to build a pupal chamber. Nematodes aggregate around the chamber and move onto the new adult of the beetle that will emerge carrying them to new host trees (Winfield, 1987). Recent discovery of the nematode causing death of *P. pinaster* trees in Portugal (Mota et al., 1999) and vectored by *Monochamus galloprovincialis* (Sousa et al., 2001) has created great concern in Europe on this quarantine organism, since earlier pest risk assessments had concluded that the nematode would survive in Europe and tree mortality would likely be important in warmer southern countries (Evans et al., 1996). Currently, research is being carried out in Europe on the chemical ecology of this beetle aimed to develop efficient traps and baits for monitoring and control of the insect vectors (Pajares et al., 2004; Ibeas et al., 2007, 2008).

Besides the occurrence of both, the nematode and the vectors of the genus *Monochamus*, several additional conditions must be fulfilled for the disease to develop in a particular area. Two main risk factors have been identified: presence of susceptible host species and prevalence of suitable temperatures; additionally, stressing conditions of trees will increase susceptibility and risk of disease development (Mamiya, 1983; Ikeda, 1996). *B. xylophilus* has been found on many conifers, particularly pine species, in many cases in its micophagous, non-pathogenic, state (i.e. in its native range in North America). On the other hand, several pines have been shown susceptible through inoculation tests, as *P. sylvestris, P. nigra* and *P. pinaster* in Europe. Other Mediterranean species as *P. halepensis* and *P. pinea* are classified as of intermediate susceptibility (Evans et al., 1996).

Temperature is the most relevant climatic factor for disease development. Analysis of disease spread in North America (on exotic susceptible pines) and Japan found that the disease never occurred on susceptible tree species if mean air temperatures of the warmest month were lower than 20°C, even though the nematode and its vector

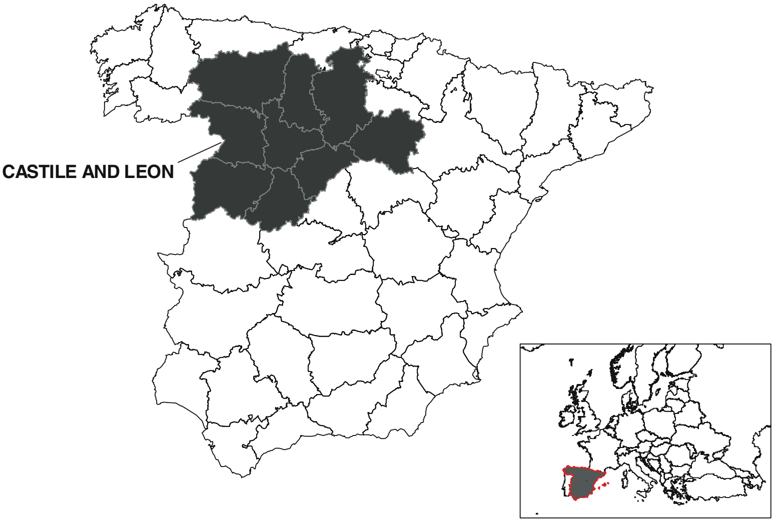


Fig. 1 Location of Castile and Leon in north-western Spain

were present (Maleck & Appleby, 1984; Rutherford et al., 1990). In Japan, most cases occurred in areas over 22°C, and in Portugal, the areas recently affected by the disease reach mean temperatures in July and August slightly above such figure. Tree stress also contributes to disease expression. Water deficits together with high temperatures increased tree susceptibility and favoured pathogen and vector development in Japan and North America (Rutherford & Webster, 1987).

In this chapter, we defined a risk rating model to establish distinct risk levels for development of pine wilt disease in Castile and Leon Autonomous Community (North western of Spain, Fig. 1) and determined: first, current forest areas were these levels occurred and second, expected changes in risk areas if global warming raised main air temperature of July by 2°C. The potential introduction of the pine wood nematode poses a serious threat to many pine forest in Europe, thus, even though the study presented is within a regional scope, the procedure here developed would be of interest for application to most of the European regions were *Monochamus galloprovinciallis* and susceptible hosts occur.

2 Material and Methods

2.1 Risk Levels

Risk levels were based on two parameters, stand composition of susceptible *Pinus* species and mean July temperatures.

Pine species in the region were classed as (Evans et al., 1996):

- Highly susceptible: *P. pinaster* (maritime pine), *P. sylvestris,* (Scots pine), *P. nigra* (Austrian pine)
- Low to moderately susceptible: *P. pinea* (stone pine), *P. halepensis* (Aleppo pine), *P. radiata* (Monterey pine)

Stands were then classed in two types based on species composition:

A: Stands with highly susceptible species as main or 2nd order species
B: Stands with low to moderately susceptible species as main or 2nd order species

Mean July temperatures: based on thermal requirements for disease expression (Rutherford & Webster, 1987; Rutherford et al., 1990) and on temperatures occurring in the affected areas of Portugal (Direçao-Geral das Florestas, 1999), four thermal risk zones were established considering mean July isotherms: >22°C, 20–22°C, 18–20°C and <18°C.

Finally, six levels of risk for disease development, from very high to very low, were defined combining both parameters (Table 1).

2.2 Mapping of Risk Levels

Application of these risk levels to the forest stands in the Autonomous Community of Castile and Leon was carried out through the Geographical Information System (GIS) Arcwiew 8.0 (Environmental System Research Institute, Inc.). Theme maps used for the determination of risk levels in each forest stand were provided digitalized by the Information System of the Environment (SIGMENA) from the Autonomous Government of Castile and Leon, except the mean July isotherms map which was digitalized and georeferenced using Autocad map (AUTODESK. ACIS, Spatial Technology Inc.) on a standard source map from the cartographic services of the Autonomous Government (Table 2). All the cartographic data bases used in this study are in UTM projection, referred to huse 30, zone T. The established geodesic system has employed the "European Datum of 1950 (Datum ED 50). Figure 2 outlines the process for generating the final theme maps showing the risk levels for all pine stands and for each pine species.

Table 1 Risk levels for pine wilt disease development

Risk level	Stand type	Thermal zone
Very high	A	>22°C
High	A	20–22°C
Moderate	A	18–20°C
Moderate-low	B	>22°C
Low	B	18–22°C
Very low	A, B	<18°C

Table 2 Theme maps used for application of risk levels to forest

Map	Spatial model	Extension	Type of file	Scale
Regional and provincial perimeters	Vectorial	SHP	Form file Arcview	1:50,000
Forest map of Spain	Vectorial	SHP	Form file Arcview	1:50,000
Localities	Vectorial	SHP	Form file Arcview	1:50,000
July isotherms	Vectorial	DXF	Interchange file Autocad	1:500,000

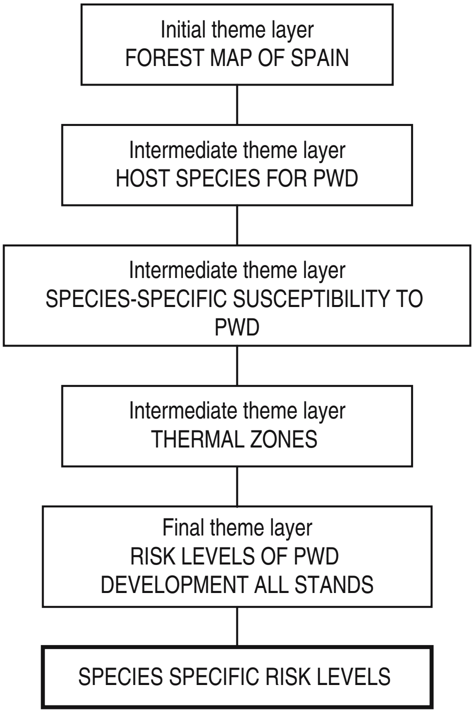


Fig. 2 Sequence for generating pine wilt disease risk maps in Castile and Leon

3 Results and Discussion

3.1 Risk Levels in Castile and Leon Pinewoods

Result of risk levels for Castile and Leon considering all pine stands together are presented in Table 3. It can be observed that almost 75% of the forest stands are currently classed within the three upper levels of risk, a result derived from the great area occupied by the disease susceptible Scots and maritime pines. Here, the bigger area, almost 400,000 ha, is associated to a moderate risk, but second in extension are the stands showing high risk (26.4% of total pine area) (Fig. 3). Only near 5% of the pinewoods exhibited a very high risk of PWD development (Table 3).

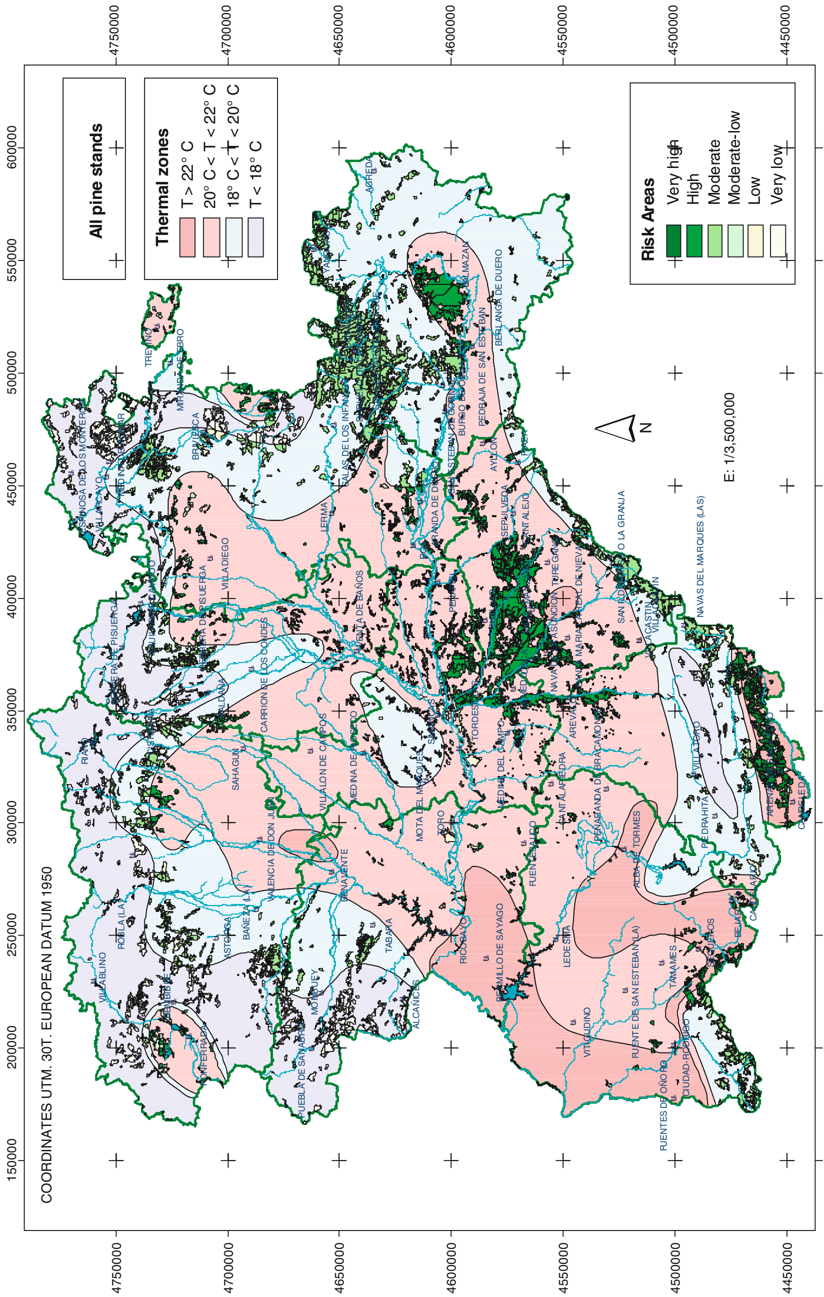


Fig. 3 Current risk levels of pine wilt disease in all pine stands in Castile and Leon

Table 3 Current and predicted pine areas (ha) associated to risk levels of pine wilt disease development in Castile and Leon

Risk level	Current area (%)	Predicted area (%)	Variation (%)
Very high	50,356 (4.9)	322,614 (31.4)	26.5
High	272,258 (26.5)	394,089 (38.4)	11.9
Moderate	394,089 (38.4)	165,049 (16.1)	−22.3
Moderate-low	51,220 (5)	76,855 (7.5)	2.5
Low	71,256 (6.9)	67,478 (6.5)	−0.4
Very low	187,686 (18.3)	780 (0.1)	−18.2

This scenario would be greatly modified if mean summer temperatures raised 2°C as consequence of climate warming. In the new situation, predicted area within the two upper risk classes encompasses near 70% of the region pinelands, resulting more than 300,000 ha under a very high risk of disease affection. Moderate risk level area would be now 3rd in rank (Table 3).

3.2 *Species-Specific Risk Levels of Disease Development*

3.2.1 Pinus pinaster

High specific susceptibility and warm distributions of most of maritime pine stands lead to a higher risk for affection of this species among the pines in Castile and Leon. In the current situation, more than 260,000 ha of *Pinus pinaster* pinewoods are subjected to a high (53.6% of total) or very high risk (10.7%) (Table 4). Stands showing higher susceptibility are located in southernmost areas of the Central Range, in the provinces of Avila and Salamanca (Alberche and Tiétar rivers, Sierra de Béjar and Peña de Francia). Vast pinelands in the plain across Segovia, Valladolid and Soria provinces present also great risk of disease development if the nematode occurred. Warming would mean a net worsening of the scenario for this species, as under simulated conditions, practically all the maritime pine stands in Castile and Leon (90%) are predicted to be within the two higher risk classes (Fig. 4, Table 4).

3.2.2 Pinus sylvestris

Most of the stands of Scots pine throughout the region are located in mountain areas below the 20°C mean summer temperature threshold. For this reason, in spite of its high intrinsic pine wilt disease susceptibility, most of pinewoods of this species are currently subjected to a moderate (56.4%) or even lower (36.5%) risk (Table 5). However, more than half of *P. sylvestris* stands are occurring in thermal zones where July temperatures are slightly below the above mentioned threshold, so a moderate increment of 2°C in July mean temperatures would notably change the prognosis for these stands, being in the new situation under a high risk of disease affection (Fig. 5, Table 5).

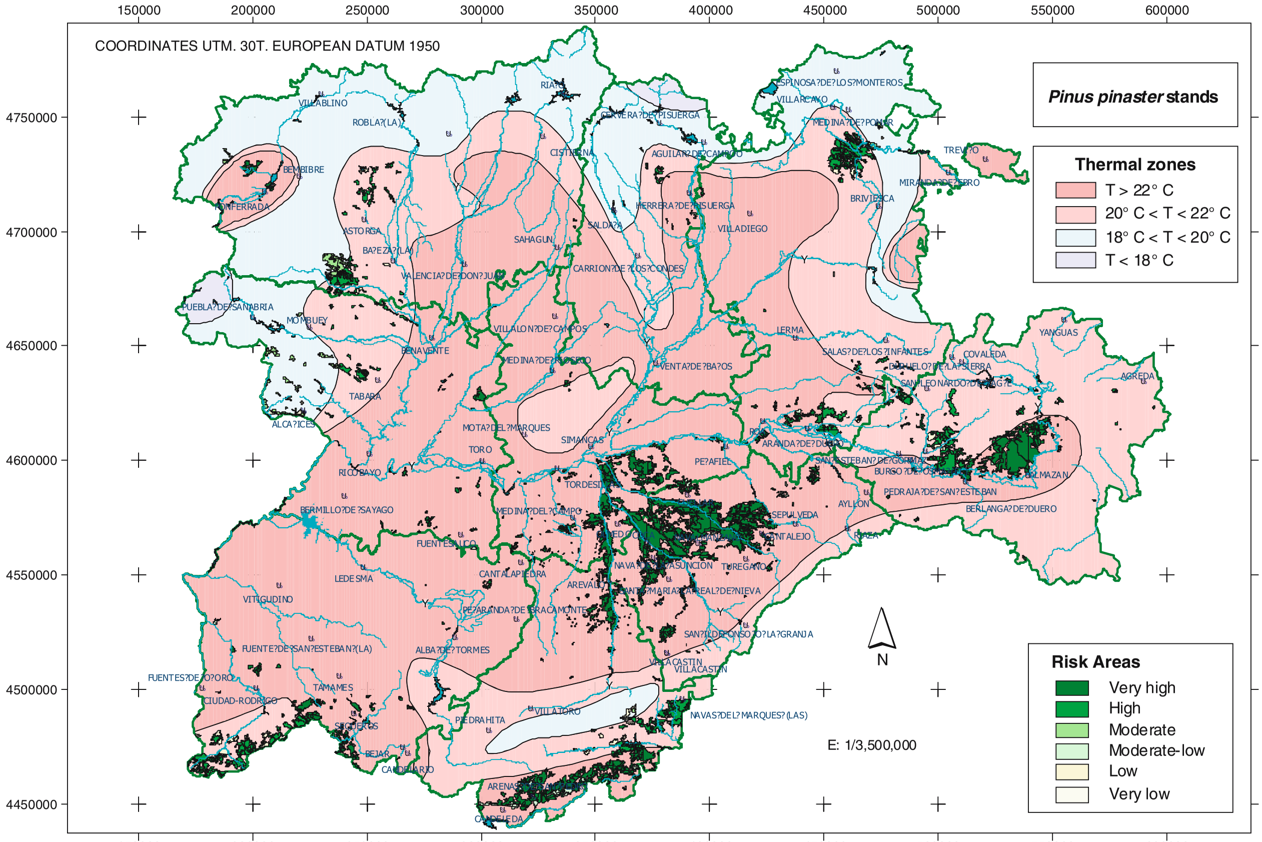


Fig. 4 Risk levels of pine wilt disease in Pinus pinaster stands in Castile and Leon. Predicted after 2°C increase of mean July temperature

Table 4 Current and predicted areas (ha) of *P. pinaster* associated to risk levels of pine wilt disease development in Castile and Leon

Risk level	Current area (%)	Predicted area (%)	Variation (%)
Very high	44,220 (10.7)	266,624 (64.3)	53.6
High	222,404 (53.6)	104,183 (25.1)	−18.5
Moderate	104,183 (25.1)	16,022 (3.9)	−21.2
Moderate-low	12,073 (2.9)	12,073 (2.9)	0
Low	12,931 (3.1)	15,719 (3.8)	0.7
Very low	18,811 (4.5)	0	−4.5

Table 5 Current and predicted areas (ha) of P. sylvestris associated to risk levels of pine wilt disease development in Castile and Leon

Risk level	Current area (%)	Predicted area (%)	Variation (%)
Very high	5,271 (1.3)	27,778 (7)	5.7
High	22,507 (5.7)	223,089 (56.5)	50.7
Moderate	223,089 (56.4)	123,140 (31)	−25.4
Moderate-low	3,029 (0.8)	3,029 (0.8)	0
Low	11,227 (2.8)	173,768 (4.5)	1.7
Very low	130,278 (32.9)	597 (0.2)	−32.7

Pinus sylvestris stands that would experience a remarkable worsening of disease risk are located in the mountains of the Central Range in Segovia and Avila (Sierras de Ayllón, Somosierra, Guadarrama and Gredos) and of the Iberian Range in Soria and Burgos (Sierra de la Demanda, Sierra Cebollera, Picos de Urbión), accounting for more than 220,000 ha of some of the most valuable pinewoods in the region.

3.2.3 Pinus nigra

The third highly susceptible species, Austrian pine, is less represented in Castile and Leon, covering some 71,000 ha region wide. Most of the *P. nigra* stands are reforestations installed in cool areas where the assessed risk level is usually moderate (51.5%), though near 20% of them are occurring in the higher risk classes (Table 6). As it happened to the species before, warming would move many of these stands across the main risk threshold, so under the simulated temperature increase, more than 70% of *P. nigra* stands are predicted to suffer a considerable risk of pine wilt disease incidence (Fig. 6). Reforested areas in high plateau piedmonts of Soria and Palencia provinces will be subjected to high risk, whereas pinelands in Segovia and Valladolid plains would change to a very high risk class.

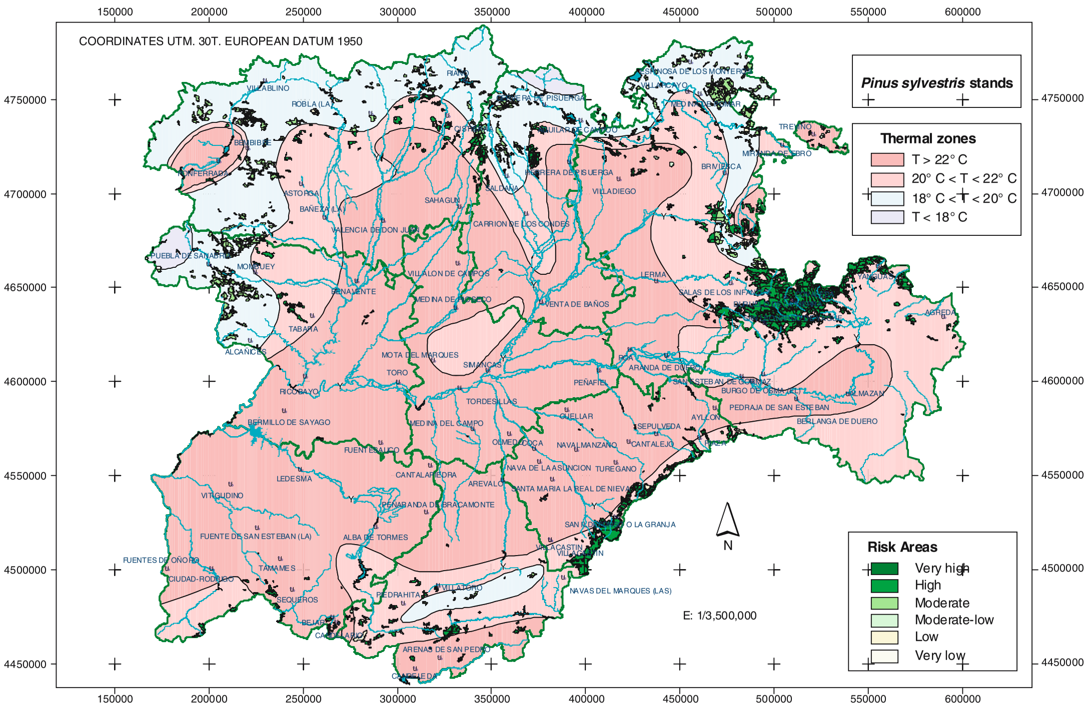


Fig. 5 Risk levels of pine wilt disease in Pinus sylvestris stands in Castile and Leon. A: current. B: predicted after 2°C increase of mean July temperature

Table 6 Current and predicted areas (ha) of *P. nigra* associated to risk levels of pine wilt disease development in Castile and Leon

Risk level	Current area (%)	Predicted area (%)	Variation (%)
Very high	92 (0.1)	14,397 (20.1)	20.1
High	14,305 (20.1)	36,720 (51.5)	31.4
Moderate	36,720 (51.5)	6,163 (8.6)	−42.9
Moderate-low	4,222 (5.9)	4,222 (5.9)	0
Low	4,797 (6.7)	9,832 (13.9)	7.2
Very low	11,209 (15.7)	0	−15.7

3.2.4 Pinus pinea, P. halepensis and P. radiata

Two other native and one introduced pine species occur in Castile and Leon, but they are of much smaller distribution: stone pine occupies slightly over 34,000 ha; Aleppo pine, mostly of reforested origin, covers 22,000 ha and Monterrey pine plantations amount to less than 9,000 ha. All these pines appear in temperate valleys and plains, but their low specific susceptibility leads them to lower risk levels. In *P. pinea*, the majority of the stands (96%) occur in the low risk class, whereas for *P. halepensis* and *P. radiata* moderately-low is the most frequently assigned level (82.5% and 51.2% respectively) (Table 7). Thus, simulation of temperature increases for these species, except for a widening of the predicted area subjected to moderately-low risk in stone pine (78% now), do not account for any remarkable variations in their risk of pine wilt disease affection.

The pine wood nematode is a very destructive organism causing a great concern in the European Union. If the pathogen is introduced in Spain, from Portugal or by shipments from other countries where it currently occurs, the concurrence of a native vector, *Monochamus galloprovincialis*, susceptible host species and warm summer temperatures will lead to a very damaging spread of pine wilt disease. In Castile and Leon, estimations of potential incidence here presented show that almost a third of over 1 million hectares of pine forest are classed as having high or very high risk for pine wilt disease development if the nematode was finally introduced. Considering each of the pines individually, *P. pinaster* appeared as the most endangered species (64.3% high or very high risk), followed by *P. nigra* (22.2%) and *P. sylvestris* (7%). Alarming as it is, this prognosis will be considerably worsened if predicted rising of temperature due to global warming will eventually occur. Simulations considering an increase of 2°C in mean air temperature of the hottest month, predicted that more than 700,000 ha of pinewoods in the region would be assigned to the two upper risk classes. Remarkable changes in risk status due to this global perturbation would include an extension of the higher risk levels to practically all maritime pine stands and, most noticeably, to almost two thirds of the valuable Scots pine forests in the Central and Iberian Ranges. It must be stressed here that a small increment of summer temperatures would greatly increase the risk for this species, and this is of particular relevance, as it might be the case for some other European regions were Scots pine is a main forest species.

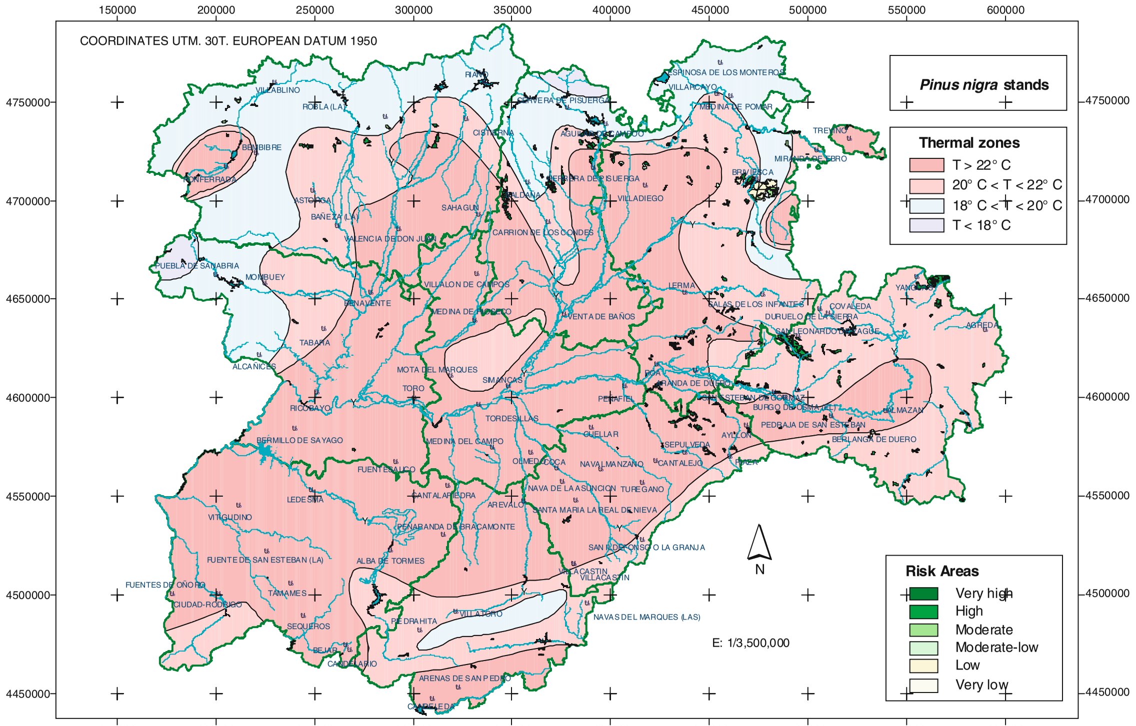


Fig. 6 Risk levels of pine wilt disease in Pinus nigra stands in Castile and Leon predicted after 2°C increase of mean July temperature

Table 7 Current and predicted areas (ha) of *P. pinea*, *P. halepensis* and *P. radiata* associated to risk levels of pine wilt disease development in Castile and Leon

Risk level	Current area (%)	Predicted area (%)	Variation (%)
Pinus pinea			
Moderate-low	1,524 (4%)	26,923 (78%)	74
Low	32,857 (96%)	7,459 (22%)	−74
Very low	0	0	0
Pinus halepensis			
Moderate-low	18,761 (82.5)	18,761 (82.5)	0
Low	3,845 (16.9)	3,977 (17.5)	0.6
Very low	132 (0.6)	0	−0.6
Pinus radiata			
Moderate-low	4,453 (51.2)	4,458 (51.2)	0
Low	568 (6.5)	4,251 (48.8)	42.3
Very low	3,683 (42.3)	0	−42.3

Our risk model has been built based on summer temperatures, species-specific susceptibility and stand composition as the sole risk parameters, however, tree stress is another important factor favouring pathogen and vector development, and thus contributing to pine wilt disease occurrence. In Japan, symptoms of dieback were more severe and disease transmission occurred faster under conditions of high temperatures and low rainfall. Great damage were observed in stressed trees after minimum rainfall and temperatures over 25°C during 55 days (Takeshita et al., 1975). Summer drought induced rapid progress of the disease and high mortality in pines inoculated with virulent PWN isolates (Mamiya, 1983) and water-stress even induced increased susceptibility to avirulent PWN (Ikeda, 1996) Problems of evaluation and quantification precluded incorporation of this parameter in our risk rating model, but we should bear in mind that climate warming in our region would involve a higher frequency of stressing conditions for trees (higher temperature and extreme drought during the summer), so risk here predicted might be likely underscored.

Acknowledgements We thank J.M. Sierra and A. Martín (Centro de Sanidad Forestal de Calabazanos, Junta de Castilla y León) and G. Sánchez (Dirección General para la Biodiversidad). Financial support was provided by a grant from Ministerio de Medio Ambiente (Dirección General para la Biodiversidad), Spain.

References

Battisti, A., Statsmy, M., Schopf, A., Roques, A., Robinet, C., Larsson, A. (2005). Expansion of geographical range in the pine processionary moth caused by increased winter temperature. *Ecological Entomology*, *15*, 2084–2094.

Direçao-Geral das Florestas (1999). *Relatorio de execuçao. Plano de erradicaçao do nematodo do pinherio. 1ª fase*. Ministerio de Agricultura, Desenvolvimento Rural e das Pescas, Lisboa.

Evans, H.F., Mc Namara, D.G., Braash, H., Chadoeuf, J., Magnusson, C. (1996). Pest Risk Analyses (PRA) for the territories of the European Union (as PRA area) on *Bursaphelenchus xylophilus* and its vectors in the genus *Monochamus. OEEP/EOPPO Bulletin*, *26*, 199–249.

Ibeas, F., Gallego, D., Díez, J.J., Pajares, J.A. (2007). An operative kairomonal lure for managing pine sawyer beetle *Monochamus galloprovincialis* (Coleoptera: Cerambycidae). *Journal of Applied Entomology*, *131*, 13–20.

Ibeas, F., Díez, J.J., Pajares, J.A. (2008). Olfactory sex attraction and mating behaviour in the pine sawyer *Monochamus galloprovincialis* (Coleoptera: Cerambycidae). *Journal of Insect Behaviour* DOI 10.1007/s10905-007-9104-1.

Ikeda, T. (1996). Responses of water-stressed *Pinus thunbergii* to Inoculation with avirulent pine wood nematode (*Bursaphelenchus xylophilus*): water relations and sylem histology. *Journal of Forest Research*, *1*, 223–226.

IPCC (2007). *Intergovernmental Panel on Climate Change*. Fourth assessment report. Available at www.ipcc.ch, last access 26/10/2007.

Maleck, R.B., Appleby, J.E. (1984). Epidemiology of pine wilt in Illinois. *Plant Disease*, 68, 180–186.

Mamiya, Y. (1983). Pathology of the pine wilt disease caused by *Bursaphelenchus xylophilus. Annual Review of Phytopathology*, *21*, 201–220.

Mamiya, Y. (1988). History of pine wilt disease in Japan. *Journal of Nematology*, *20*, 219–226.

Mota, M.M., Braasch, H., Bravo, M.A., Penas, A.C., Burgermeister, W., Metge, K., Sousa, E. (1999). First report of *Bursaphelenchus xylophilus* in Portugal and in Europe. *Nematology*, *1*, 727–734.

Pajares, J.A., Ibeas, F., Díez, J.J., Gallego, D. (2004). Attractive responses by *Monochamus galloprovincialis* (Col., Cerambycidae) to host and bark beetle semiochemicals. *Journal of Applied Entomology*, *128*, 633–638.

Rutherford, T.A., Webster, J.M. (1987). Distribution of pine wilt disease with respect to temperature in North America, Japan and Europe. *Canadian Journal of Forest Research*, *17*, 1050–1059.

Rutherford, T.A., Mmamiya, Y., Webster, J.M. (1990). Nematode-induced pine wilt disease: factors influencing its occurrence and distribution. *Forest Science*, *36*, 145–155.

Sousa, E., Bravo, M.A., Pires, J.,. Naves, P., Penas, A.C., Bonifacio, L., Mota, M.M. (2001). *Bursaphelenchus xylophilus* (Nematoda: Aphelenchoididae) associated with *Monochamus galloprovincialis* (Coleoptera: Cerambycidae) in Portugal. *Nematology*, *3*, 89–91.

Takeshita, K., Hagihara, Y., Ogawa, S. (1975). Environmental analysis to pine damage in Western Japan (Japanese with English summary). *Bulletin of Fukuoka Forest Experimental Station*, *24*, 1–45.

Williams, D.W., Liebhold A.M. (2002). Climate change and the outbreak range of two North American bark beetles. *Agricultural and Forest Entomology*, *4*, 87–99.

Winfield, M.-J. (1987). A comparison of mycophagous and phytophagous phases in the pine wood nematode. In: Winfield, M.-J. (ed) *Pathogenicity of the Pine Wood Nematode*. APS Press, St. Paul, Minnesota.

Dynamics of Soil Carbon in Primary and Secondary Tropical Forests in Colombia

F. H. Moreno[1,2]* and S. F. Oberbauer[2]

1 Introduction

Atmospheric CO_2 concentration has been increasing since the beginning of industrial revolution (Keeling, 1997; Wuebbles et al., 1999) and is expected to continue growing at progressively higher rates in coming years, with significant consequences on global climate and ecosystem dynamics (IPCC, 1996; Schimel et al., 1994). The potential sinks of this atmospheric CO_2 are terrestrial ecosystems and oceans (Fan et al., 1998). Tropical forests could play an important role in the absorption of excess CO_2 because they are the largest pool of organic C among terrestrial ecosystems. Tropical forests cover only 17% of terrestrial surface of the planet and store about 42% of C contained in biomass, 59% of C in forests (Dixon et al., 1994), and 27% of C in soils; they also account for 36% of all terrestrial net primary productivity NPP (Melillo et al., 1993; Phillips et al., 1998).

Tropical deforestation is a critical global environmental problem because the 15.4 million hectares cleared each year mainly for agriculture and grazing (FAO, 1993; Nepstad et al., 1991) are a disproportionately large source of atmospheric carbon (1.1–1.6 Pg C; $1\,\mathrm{Pg} = 10^{15}$ g) (Batjes & Sombroek, 1997). Simultaneously, secondary forests are extensive in the tropics because of the increasing abandonment of many lands in this region. In 1990 they accounted for about 40% of the tropical forest area (Brown, 1990), with the potential of being significant carbon sinks (Silver et al., 2000)

Forest carbon cycles involve inputs, storage, transfers and outputs of carbon between two large pools: vegetation and soils. Main processes in vegetation are photosynthesis, growth of above and belowground plant organs, and production of dead material. Carbon in plant biomass goes back to the atmosphere through

---

[1] Universidad Nacional de Colombia, Sede Medellín, Departamento de Ciencias Forestales, A.A. 568 Medellín, Colombia, *fhmoreno@unalmed.edu.co

[2] Florida International University, Department of Biology, University Park, Miami, FL 33199 oberbaue@fiu.edu

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

respiration of living tissues and decomposition of standing dead organs. Processes in soils involve input of dead organic mater, which mostly accumulates on the forest floor and in the soil profile. This material is decomposed by heterotrophic respiration, which returns to the atmosphere a significant proportion of the carbon fixed by photosynthesis (Schlesinger, 1997).

Soils play a central role in the dynamics of carbon in the biosphere because a small change in the rate of soil carbon efflux can profoundly affect the global carbon cycle and the amount of carbon stored in the atmosphere, with important effects on the global climate and the biosphere. Soil respiration releases from 50 to 70 Pg C annually (Boone et al., 1998), which is equivalent to 10–15 times the amount released yearly by fossil fuel consumption (Schlesinger, 1997). Furthermore, about 30% of current carbon emissions by land use change (estimated at 2.0 ± 0.8 Gt C year^{-1}) comes from loss of soil organic carbon (SOC) (Batjes & Sombroek, 1997; Houghton, 1991, 1999).

Soils also contain the largest carbon stock in terrestrial ecosystems. The first meter of soil has about 1,500 Pg of carbon, an amount equaling three times that stored in aboveground vegetation and twice that of the atmosphere (Batjes & Sombroek, 1997; Schlesinger, 1997). Soils of tropical forests are among the largest carbon reservoirs of world's soils (Jobbagy & Jackson, 2000). They receive the largest organic matter inputs because of the greater plant productivity in these ecosystems (Landsberg & Gower, 1997; Raich & Schlesinger, 1992; Saugier et al., 2001) and also have the largest soil respiration rates, mainly because of the high temperatures (Jenkinson, et al., 1991; Raich & Schlesinger, 1992).

Soil organic matter (SOM) results from the balance between two major carbon fluxes: detrital inputs from above plus belowground sources (litterfall and root mortality) minus carbon outputs from heterotrophic respiration by the decomposition of SOM (Fig. 1), which is believed to be the main path of carbon return to the atmosphere (Hanson et al., 2000). In forests, other fluxes of carbon such as precipitation, dry deposition, leaching, runoff, erosion, etc. are small in comparison with

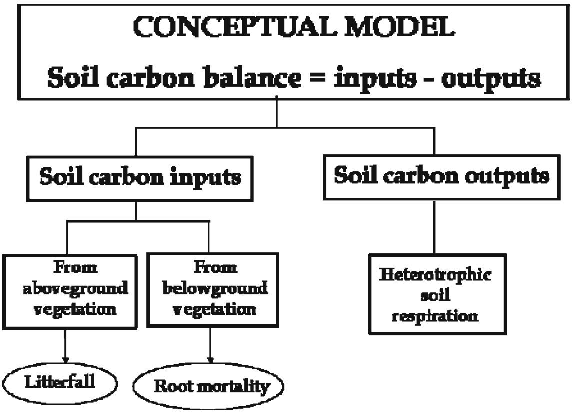


Fig. 1 Conceptual model of soil carbon balance

the major fluxes and usually have not been measured in tropical forests (Clark et al., 2001; Sierra et al., 2007).

Under a nearly steady-state condition, mean annual inputs are approximately equal to outputs by respiration of decomposers (Hanson et al., 2000; Raich & Nadelhoffer, 1989), then,

$$HR \approx Pa + Pb \tag{1}$$

Where HR is the heterotrophic respiration, Pa and Pb are production of above and belowground litter. The inclusion of woody detritus in the estimation of Pa is under debate. Because of the small size of collars used to measure total soil and heterotrophic respiration (10 cm diameter), they did not include CO_2 contributions from large items such as palm fronds and woody detritus on soil surface. Though some carbon from woody detritus is probably incorporated into the soil, most of this carbon is directly released to the atmosphere without first entering the soil (Raich & Nadelhoffer, 1989). Therefore, the error due to ignoring carbon fluxes from decomposition of woody debris is probably small. The soil carbon balance from Eq. (1) is then

$$SCB = Inputs - outputs = (Pa + Pb) - HR \tag{2}$$

Despite the tremendous importance of tropical forests for the global carbon cycle and the critical role of soil processes in biosphere-level carbon fluxes, the dynamics of soil carbon in tropical forests are still poorly understood. The key to understanding the carbon balance in forests and to predicting the effect of land use changes on carbon pools is to study the processes governing the soil carbon dynamics. The problem examined in this paper involves the balance inputs-outputs of soil carbon and its effect on carbon dynamics in soils of primary tropical forests and young secondary forests developed in degraded areas after several decades of unmanaged pastures in a region of the Colombian Andes.

2 Materials and Methods

2.1 Site Description

The study was carried out in the middle watershed of the Porce River (6° 45′37″ N, 75° 06′28″ W), located in the northern region of the Central Cordillera of the Andes in Colombia. The study area comprises a 5,000-ha land tract used to protect the reservoir of the Hydroelectric Project Porce II (Fig. 2). The average annual rainfall in the center of the study area between 1990 and 2002 was 2,078 mm (SD = 243.8 mm), with a dry season (monthly rainfall less than 100 mm) from December to February. The wet season extends from March to November, with a short rainfall decrease from June to August. Elevation of the study plots ranged from 1,000 to 1,300 m a.s.l. where the estimated average temperature ranged from 21.3°C to 23.1°C, depending on elevation. Environmental conditions correspond to the Tropical Premontane Moist Forest life zone (Holdridge, 1978). Parent materials of soils are igneous

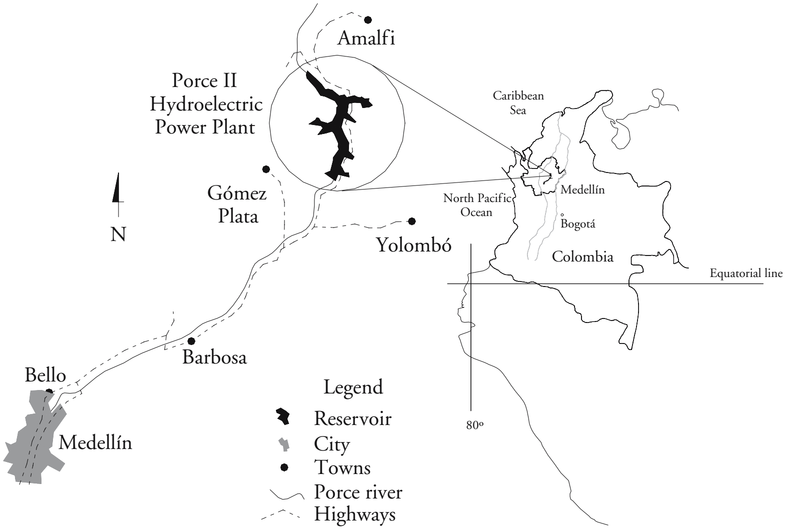


Fig. 2 Location of the study area

granitic rocks composed mainly by quartz-diorite. They are well-drained, acidic (pH = 5), with low natural fertility, and are classified as Ustoxic Dystropept (Jaramillo, 1989).

From 75 permanent plots 0.05 ha each (20 × 25 m) established in secondary forests (SF) and 33 plots 0.1 ha each (20 × 50 m) in primary forests (PF), we selected three of these plots in 10 year-old SF and three in PF, all located on steep topography (slope rank of 20°–35°). Sampling plots were chosen with soil and terrain conditions as similar as possible to allow valid comparisons between forest types. Previous land use in plots of SF was degraded pasture which was abandoned to natural succession upon land purchase in the late 1980s and early 1990s. Degraded pasture is the most common and extensive land use type in the Colombian Andes. Colonization of this region occurred mostly in the 18th and 19th centuries and pastures in this area were more than 50 year-old.

2.2 Litterfall

Production of plant aboveground fine litter was evaluated in 10 litter traps randomly placed inside each plot (60 traps in total). Each trap had a circular surface area of 0.5 m^2, was 56 cm deep, and stood on three legs 70 cm above the soil surface. Traps were built using wire rings covered with fine mesh fabric (muslin, 0.5 mm mesh size) to avoid both water accumulation and loss of material. Fine litterfall was collected about every 2 weeks during 32 months from February 2001 to September 2003. During this period a total of 57 collections of fine litter were made. Litter collected inside traps was placed into individual, pre-labeled polyethylene bags, then sealed and transported to the laboratory for drying to a constant weight at 70°C. Carbon content was determined in a NC 1500 Carlo Erba CHN analyzer (Carlo Erba Instruments, Milan, Italy).

2.3 Production of Fine Root Debris

Root production was estimated by the ingrowth core method (Persson, 1990; Vogt et al., 1998). Four subplots 2 × 2 m each were systematically located at a distance of 5 m from each other along the center of each plot and parallel to the contour, starting at 2.5 m from the border of the plot. At every sampling date a random point was chosen inside each subplot and a soil sample down to 30 cm-depth was taken. We used the same holes opened for the extraction of soil samples in a study of root mass by the sequential core method (Persson, 1990; Vogt et al., 1998), which were immediately filled with previously sieved (0.54 mesh size) and root-free soil from the same plot and depth range. Harvests were done approximately every 45 days and simultaneous with the sequential core samples (12 harvests over 535 days between 2001 and 2003).

Samples were taken with a 77884 AMS Core Sampling Mini Kit (Forestry Suppliers Inc., Jackson, MS), which has an internal diameter of 4.7 and 35 cm length. A hammer attached to the upper part of the core sampler facilitated penetration of the sampler into the soil. The soil surface was brushed clean of aboveground litter before sampling. Separation of fine roots from soil was performed with a custom-made root elutriator adapted from the design of Caldwell and Virginia (1989). After washing in the elutriator, final sorting was done by hand.

We evaluated root mortality in these forests by combining estimated live root mass from sequential core sampling and root production obtained with the ingrowth technique. Mortality of live fine roots (Mg ha^{-1}) between time 1 and time 2 ($M_{(t1,\, t2)}$) was estimated as the difference between the expected biomass of live root at time 2 (i.e. the sum of live roots at time 1 plus root production in the period) and the observed biomass of live roots at time 2,

$$M(t_1, t_2) = (Bt_1 + P(t_1, t_2)) - Bt_2 = (Bt_1 - Bt_2) + P(t_1, t_2)$$

$$M(t_1, t_2) = \Delta B(t_1, t_2) + P(t_1, t_2) = P(t_1, t_2) - \Delta B(t_2, t_1) \qquad (3)$$

Where P is the production of fine roots between time 1 and time 2, and ΔB is the statistically significant change in live fine root biomass between time 1 and time 2 (Persson, 1990; Vogt et al., 1998).

In an accompanying study, we sampled total fine root mass in four points per plot up to 1 m-depth (data not shown) and found that fine roots in the first 30 cm of soil represented a proportion from 0.61 to 0.85 of the fine root inventory up to 1 m-depth. We used these proportions to estimate the yearly root mortality up to 1 m depth assuming a similar trend of root dynamics below 30 cm depth.

2.4 Total and Heterotrophic Soil Respiration

Two paired subplots were established adjacent to each plot, one for the study of total soil respiration and the other for the study of heterotrophic respiration. Because other studies were in progress within the same plots, subplots were located outside them to avoid trampling. The trenching plot approach was used for the study of heterotrophic respiration (Boone et al., 1998; Hanson et al., 2000; Kelting et al., 1998). A trench 95 cm depth was dug along the perimeter of the subplot (2 × 2 m) and zinc flashing 0.92 cm width was placed in the trench surrounding the subplot; finally the soil was carefully put back into the trench, minimizing disturbance inside the subplot by the trenching treatment. Zinc flashing provided a physical barrier to root ingrowth into the subplot. Close to each trenching subplot a paired untreated subplot was established for monitoring total soil respiration. All vegetation was periodically removed from trenching subplots to prevent the growth of living roots, which could release CO_2 by respiration, as well as living plants, which could absorb CO_2 by photosynthesis. To prevent biases in the location of the subplots, a protocol was followed to randomly locate subplots. Five 10.5 cm diameter and 4.5 cm height PVC collars were inserted 1.5 cm into the soil surface in each subplot (a set of 60 PVC collars in the whole experiment). All vegetation inside the collars was also removed.

Soil carbon efflux was measured approximately every 2 weeks, starting 2 weeks after experimental setup using the flow-through enclosure approach (Holland et al., 1999). Measurements were performed with a LI-COR soil respiration chamber (Li-6000-09, LI-COR Inc., Lincoln NE USA) connected to a LI-COR portable photosynthesis system (Li-6200), which measures CO_2 concentration with an infrared gas analyzer (IRGA). The fieldwork was carried out in two stages because of equipment failure in the middle of the experiment. Sixteen samplings were carried out in the first stage, over 278 days between 2001 and 2002 and twelve samplings in the second stage over 168 days in 2003. Soil carbon efflux was measured on the 60 collars at every sampling with three observations per collar. To prevent artifacts on soil carbon efflux produced by the alteration of the natural concentration gradient of CO_2 within the soil profile (Davidson et al., 2002), CO_2 concentration inside the soil respiration chamber was kept within ±40 ppm from the exterior air. Recorded data of soil respiration were checked immediately after measurement. If the three observations differed by more than 25%, the measurements were repeated.

Soil temperature and water content in the 0–20 cm-depth layer as well as air temperature were recorded with each soil respiration measurement. Temperature was measured with the thermistors of the soil respiration chamber and soil water (% volume) with a CS-615 reflectometer (Campbell Scientific, Inc. Logan, Utah) connected to a portable multimeter (O'Brien & Oberbauer, 2001). Data of moisture and temperature were taken concurrently along with soil respiration measurements during daytime hours (from 600 to 1800). Soil temperature and soil water content at 10 and 50 cm depth for each forest type were also recorded in two automated micrometeorological stations, one of them located in PF and the other in SF. Temperatures were measured with cooper-constantan thermocouples and soil water content with

CS-615 reflectometers (Campbell Scientific, Inc. Logan, Utah) connected to a CR 10X data logger (Campbell Scientific, Inc. Logan, Utah). Sensor readings were conducted every minute and data storage every half hour.

3 Results and Discussion

Carbon inputs from above and belowground sources were lower in soils of SF and equaled about 60% of that in PF. However, differences between forest types for root mortality were non significant (Fig. 3) probably because the variation of root mortality among plots of the same forest type was high. This spatial variability suggests the need of large sample sizes for studying fine root dynamics. Greater litterfall in PF than in SF has been usually reported (de Camargo et al., 1999; Raich, 1983) though greater litterfall in SF developed from small disturbance has also been reported (Rodrigues et al., 2001). The capacity of secondary forests to produce new biomass seems to depend on the degradation produced by the previous land use (Silver et al., 2000). The low values of carbon inputs in our sites of SF as compared to PF are a clear indicator of land degradation after decades of unmanaged grazing.

Pearson correlation coefficients between soil respiration and soil temperature were positive. Because soil temperatures during daytime hours (when soil respiration

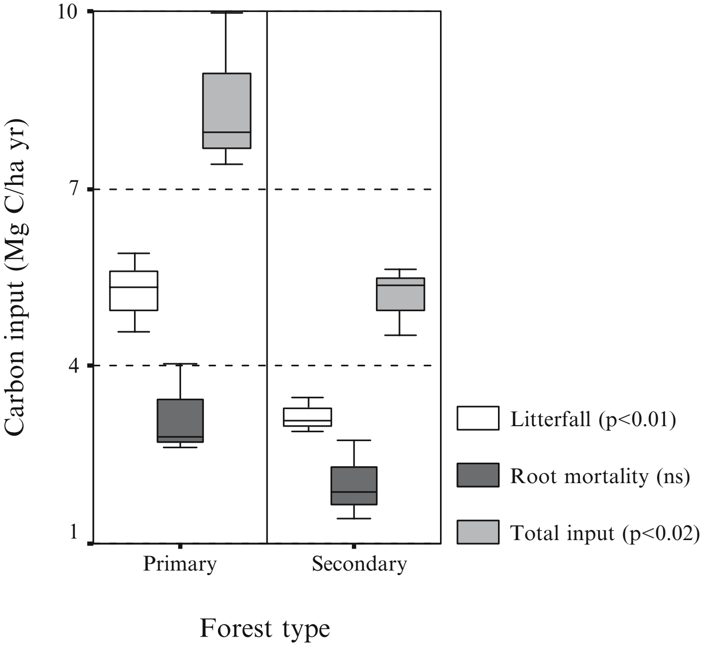


Fig. 3 Box plot of soil carbon inputs from litterfall and mortality of fine roots in primary and secondary forests. Differences between forest types were significant for litterfall ($p < 0.01$) and total carbon inputs ($p < 0.02$) and non siginificant (ns) for root mortality

measurements were done) were higher than mean daily soil temperature, gross results of soil respiration were overestimated. Correction factors estimated from linear equations of soil respiration vs. temperature for each forest type and the data of hourly temperatures from the automated weather stations were used for correcting estimations of total soil respiration (Moreno, 2004). The average estimated values for the correction factors scaled to a yearly and hectare basis were equivalent to 2.81 and 0.764 Mg CO_2-C ha^{-1} $year^{-1}$, for PF and SF, respectively (Table 1).

Soil respiration increased with soil water content in dry soils (positive correlation) up to a maximum and then decreased with further increases in soil water (negative correlation). Maximum soil CO_2 efflux occurred at soil water content of 30.0% and 31.7% (v/v) for PF and SF, respectively. The mean soil water content in the trenching subplots was 44.9% (2.73% SE), a value significantly higher (about 9%, $p < 0.01$) than that of soils of the untreated subplots. Correlation of heterotrophic respiration vs. soil water content in the trenching treatment was negative, which suggests that C efflux decreased as soil water content increased. Therefore, measured heterotrophic respiration from trenching subplots likely underestimated the respiration rates occurring in undisturbed soils where soil water content was lower. With the data of soil water regime under undisturbed conditions, (Moreno, 2004) estimated a corrected heterotrophic respiration. These conditions corresponded to those in the paired untreated subplot of the same forest plot for each sampling date where soil respiration was measured (Table 1).

One more correction term was considered for an unbiased estimation of heterotrophic respiration: the CO_2 contribution by decomposition of live roots killed by the trenching treatment (Lee et al., 2003). This correction factor was estimated from the average fine root biomass down to 1 m-depth in the soil profile per plot and the decomposition factor of fine roots K (Moreno, 2004) (Table 1). After all corrections,

Table 1 Total soil and heterotrophic respiration and correction factors in primary and secondary forests (all figures in Mg C ha^{-1} $year^{-1}$). SR_{uncorr} and $SR_{T\ corr}$ are total soil respiration (uncorrected and temperature-corrected, resp.); HR_{uncorr}, $HR_{sw\ corr}$ and $HR_{swlr\ corr}$ are heterotrophic respiration (uncorrected, soil water-corrected, and soil water-live roots corrected, respectively); SWC is soil water content; DLRK is the estimated decomposition of live roots killed in trenching plots

	Primary forests		Secondary forests	
Variable	Average	SD	Average	SD
Untreated subplots				
SR_{uncorr} (Mg ha^{-1} $year^{-1}$)	18.83	7.85	14.56	3.97
$SR_{T\ corr}$ (Mg ha^{-1} $year^{-1}$)	16.02	6.43	13.80	3.71
SWC (% v/v)	34.17	4.37	37.57	5.90
Trenching subplots				
HR_{uncorr} (Mg ha^{-1} $year^{-1}$)	6.92	0.48	6.44	0.80
SWC (% v/v)	44.65	2.78	45.38	4.92
$HR_{sw\ corr}$ (Mg ha^{-1} $year^{-1}$)	8.80	0.41	8.47	1.38
DLRK (Mg ha^{-1} $year^{-1}$)	0.81	0.09	0.71	0.14
$HR_{swlr\ corr}$N (Mg ha^{-1} $year^{-1}$)	7.98	0.50	7.77	1.39

the estimated heterotrophic respiration was very close between PF and SF (7.981 and 7.766 Mg CO_2-C ha^{-1} year^{-1}, respectively. Table 1).

The estimated mean soil carbon balance (SCB) according to Eq. 2 was 0.66 (2.50) and −2.61 (1.70) Mg ha^{-1} year^{-1} for PF and SF, respectively (Fig. 4). Standard deviations were large, which suggests that considerable spatial variation of soil carbon processes occurs among plots of the same forest type, which could overcome the apparent differences in mean soil carbon balance between primary and secondary forests ($p > 0.13$, Fig. 3).

Under steady-state the SCB is expected to be zero (Raich & Nadelhoffer, 1989). We tested this null hypothesis (SCB = 0) for each forest type and found non significant differences in both cases. Therefore, we were unable to reject the null hypothesis despite the negative balance in soils of SF, which suggest that they are less prone to accumulate carbon or even that they actually could be a net carbon source. From these results we conclude that 2 years of sampling and the six plots studied here are not enough for detecting a clear signal in the soil carbon balance produced by each forest type. Probably this situation occurred because the calculation of SCB involves additions and subtractions of large fluxes, which are prone to large spatial and temporal variability and errors.

Although a site may be near steady-state, inter annual variation in these fluxes occurs due to several factors, climate included (Davidson et al., 2002). For example, most of soil respiration data of this study were taken in 2002, which was an El Niño year, with an average rainfall of 1435.3 mm, 491.4 mm below the mean for the period 1990–2002 (1,926.7 mm). Average air temperature was 23.86°C, 0.61°C above the 1999–2003 average (23.25°C). Because the positive correlation of soil respiration both with temperature and soil water content under dry conditions in the studied forests reported above, the warmer and drier conditions produced by El Niño year probably increased soil respiration.

Given the uncertainty in the fluxes reported above, we did a cross-checking with data of soil carbon stocks of sites which are comparable in soil characteristics. These data were taken in an accompanying study down to 4 m depth in a chronosequence of degraded pastures, 10 year-old SF, and PF in the same area (Moreno, 2004). Soils of secondary forests had an average of 21.3 Mg ha^{-1} more carbon than soils of degraded pastures (Table 2), suggesting that they have already started to recover the soil carbon stocks lost under degraded pastures at a mean rate of 2.1 Mg C ha^{-1} year^{-1} (21.3 Mg C ha^{-1} divided by 10 years, the age of secondary forests), which also has been reported by other authors (de Koning et al., 2003; Silver et al., 2000), though at lower rates than those reported here. These contradictory results for soil carbon balance in SF estimated with two independent methods, confirm the need for long-term data of soil carbon fluxes.

Because of the lower organic matter inputs in soils of secondary forests, we expected lower figures of heterotrophic respiration in soils of this forest type. However, averages were almost identical between them (mean heterotrophic respiration in secondary forests was 97% of that in primary forests, Table 1). Therefore, production of plant detritus seems to be controlling the net balance of organic carbon in the soils, which suggests that SCB tends to be lower in SF.

Table 2 Basal area and carbon stocks in biomass (above and belowground) and soils (0–400 cm depth) in primary and secondary forests

	Primary forests		Secondary forests		Degraded pastures	
Variable	Mean	Standard deviation	Mean	Standard deviation	Mean	Standard deviation
Basal area (m^2 ha^{-1})	36.30	2.02	16.65	1.89	–	–
Biomass Carbon (Mg ha^{-1})	81.94	8.39	28.32	5.67	N/A	N/A
Soil organic Carbon (Mg ha^{-1})	227.93	28.12	192.52	25.55	171.18	26.98

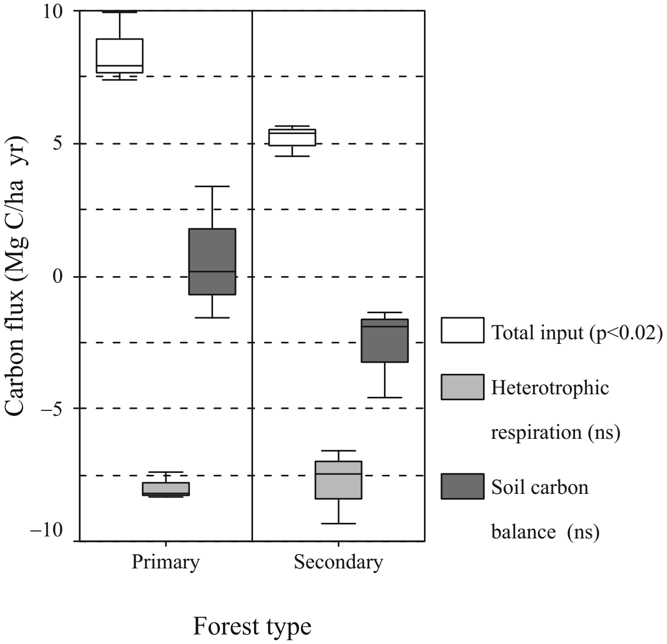


Fig. 4 Box plot of soil carbon inputs and outputs (heterotrophic respiration) and soil carbon balance in primary and secondary forests. Differences between forest types were significant for total carbon inputs ($p < 0.02$) and non siginificant (ns) for heterotrophic respiration and soil carbon balance

Consequently, we hypothesize that a similar soil carbon balance between PF and SF will occur once litterfall and fine root mortality in secondary forests reach comparable levels to those of primary forests, which is closely linked to net primary productivity (Clark et al., 2001).

Considering that SF studied here are probably degraded successions developing after several decades of poorly-managed pastures and that inputs to their soils were already 60% of those in PF after 10 years of land abandonment, a number of years will be required before they reach similar input levels. Due to the close similarity of heterotrophic respiration between PF and SF, soil carbon balance seems to basically depend on inputs from net primary productivity. Management of secondary forests and restoration efforts directed to accelerate growth rates and productivity could improve soil carbon balances and increase the capacity of soils for sequestering carbon in shorter time periods. This aspect has a remarkable importance because shows a path commonly overlooked through restoration programs in tropical secondary forests with tremendous potential for carbon sequestration. Similarly, factors associated with the increase of primary productivity, such as N deposition, CO_2 fertilization, and in some cases the changing climate (Geider et al., 2001; Nemani et al., 2003), could contribute to positive balances of soil carbon, even though plant biomass remains unchanged.

4 Conclusions

Total soil inputs of organic carbon from above and belowground sources were substantially lower in 10 year-old secondary forests developed form degraded pastures and equaled to 60% of that in comparable primary forests. However, outputs of mineral carbon through heterotrophic soil respiration were very similar; in secondary forests they equaled 97% of those in primary forests. As a consequence, the balance of carbon in the soil seems to basically depend on net primary productivity. Despite the differences in soil carbon balance between primary and secondary forests, they didn't have statistical significance probably because of the spatial and temporal variability of the fluxes studied. Management of secondary forests to accelerate growth rates and structural complexity has the potential of improving net primary productivity and carbon sequestration in soils.

Despite soil carbon balances don't show that soils of secondary forests are gaining carbon, data of soil carbon stocks down to 4 m depth in a chronosequence of degraded pastures, secondary forests and primary forests, showed that soils of secondary forests have already started to recover soil carbon stocks lost under degraded pastures at a mean rate of 2.1 Mg C ha^{-1}. This discrepancy can be at least partially explained by the increased heterotrophic soil respiration as a consequence of the drier and warmer conditions produced by a moderate El Niño event during the period of soil respiration measurements. Because interannual variation in all processes is expected, more definite conclusions require long term assessment of carbon stocks, inputs, outputs, and balances in forest soils such as this one shown here. Overall, our results showed that soil carbon balance was associated with successional status as well as with environmental conditions. Consequently, deforestation and global warming will likely strongly affect global budgets of soil carbon in the tropics.

Acknowledgements This research was primarily funded by the research program at the Universidad Nacional de Colombia, Sede Medellín (DIME) through the projects "Dinámica del carbono del suelo" and "Productividad primaria neta de las coberturas vegetales del cañón del río Porce". Additional support came from the Florida International University Tropical Biology Program. The Department of Forest Science at the Universidad Nacional de Colombia, Sede Medellín provided local transportation to the study area and laboratory facilities for sample preparation and processing. Special thanks to Empresas Públicas de Medellín (EPM) for the permission to work on their lands, lodging, logistic support during the fieldwork, and meteorological data. We are grateful to Marcela Aristizábal, Sandra Agudelo, Paola Usme, Andrés Sierra, Wilson Lara, Juan Manuel Vélez, León Montes, and Natalia Estrada, who participated in the field and lab work, as well as many fieldworkers whose help was invaluable during fieldwork. Jorge Ignacio del Valle and Sergio Alonso Orrego shared their data on aboveground vegetation and gave us important logistic and emotional support.

References

Batjes, N. H., Sombroek, W. G. (1997). Possibilities for carbon sequestration in tropical and subtropical soils. *Global Change Biology*, *3*(2), 161–173.

Boone, R. D., Nadelhoffer, K. J., Canary, J. D., Kaye, J. P. (1998). Roots exert a strong influence on the temperature sensitivity of soil respiration. *Nature*, *396*(6711), 570–572.

Brown, S. (1990). Tropical secondary forests. *Journal of Tropical Ecology*, *6*, 1–32.

Caldwell, M. M., Virginia, R. A. (1989). Root systems. In R. W. Pearcy, J. R. Ehleringer, H. A. Mooney & P. W. Rundel (Eds.), Plant physiological ecology. Field methods and instrumentation (pp. 367–398). London: Chapman & Hall.

Clark, D. A., Brown, S., Kicklighter, D. W., Chambers, J. Q., Thomlinson, J. R., Ni, J. (2001). Measuring net primary production in forests: concepts and field methods. *Ecological Applications*, *11*(2), 356–370.

Davidson, E. A., Savage, K., Bolstad, P., Clark, D. A., Curtis, P. S., Ellsworth, D. S., et al. (2002). Belowground carbon allocation in forests estimated from litterfall and IRGA-based soil respiration measurements. *Agricultural and Forest Meteorology*, *113*(1–4), 39–51.

Davidson, E. A., Savage, K., Verchot, L. V., Navarro, R. (2002). Minimizing artifacts and biases in chamber-based measurements of soil respiration. *Agricultural and Forest Meteorology*, *113*(1–4), 21–37.

de Camargo, P. B., Trumbore, S. E., Martinelli, L. A., Davidson, E. A., Nepstad, D. C., Victoria, R. L. (1999). Soil carbon dynamics in regrowing forest of eastern Amazonia. *Global Change Biology*, *5*(6), 693–702.

de Koning, G. H. J., Veldkamp, E., Lopez-Ulloa, M. (2003). Quantification of carbon sequestration in soils following pasture to forest conversion in northwestern Ecuador. *Global Biogeochemical Cycles*, 17(4), 1098, doi:1010.1029/2003GB002099.

Dixon, R. K., Brown, S., Houghton, R. A., Solomon, A. M., Trexler, M. C., Wisniewski, J. (1994). Carbon pools and flux of global forest ecosystems. *Science*, *263*(5144), 185–190.

Fan, S., Gloor, M., Mahlman, J., Pacala, S., Sarmiento, J., Takahashi, T., et al. (1998). A large terrestrial carbon sink in North America implied by atmospheric and oceanic carbon dioxide data and models. *Science*, *282*(5388), 442–446.

FAO (1993). Summary of the final report of the Forest Resources Assessment 1990 for The Tropical World. Rome: FAO.

Geider, R. J., Delucia, E. H., Falkowski, P. G., Finzi, A. C., Grime, J. P., Grace, J., et al. (2001). Primary productivity of planet earth: biological determinants and physical constraints in terrestrial and aquatic habitats. *Global Change Biology*, *7*(8), 849–882.

Hanson, P. J., Edwards, N. T., Garten, C. T., Andrews, J. A. (2000). Separating root and soil microbial contributions to soil respiration: a review of methods and observations. *Biogeochemistry*, *48*(1), 115–146.

Holdridge, L. R. (1978). Ecología Basada en Zonas de Vida. San José: IICA.

Holland, E. A., Robertson, G. P., Greenberg, J., Groffman, P. M., Boone, R. D., Gosz, J. R. (1999). Soil CO_2, N_2O, and CH_4 exchange. In G. P. Robertson, D. C. Coleman, C. S. Bledsoe & P. Sollins (Eds.), Standard soil methods for long term ecological research (pp. 185–201). New York: Oxford University Press.

Houghton, R. A. (1991). Tropical deforestation and atmospheric carbon dioxide. *Climatic change*, *19*, 99–118.

Houghton, R. A. (1999). The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. *Tellus Series B-Chemical and Physical Meteorology*, *51*(2), 298–313.

IPCC (1996). Climate change 1995: the science of climate change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

Jaramillo, D. F. (1989). Estudio General de Suelos, erosión y Uso Potencial Agropecuario para los Proyectos Hidroeléctricos Porce II y Porce III (pp. 88). Medellín, Colombia: Empresas Públicas de Medellín.

Jenkinson, D. S., Adams, D. E., Wild, A. (1991). Model estimates of CO_2 emissions from soil in response to global warming. *Nature*, *351*(6324), 304–306.

Jobbagy, E. G., Jackson, R. B. (2000). Belowground processes and global change - the vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, *10*(2), 423.

Keeling, C. D. (1997). Climate change and carbon dioxide: an introduction. *Proceedings of the National Academy of Sciences of the United States of America*, *94(*16), 8273.

Kelting, D. L., Burger, J. A., Edwards, G. S. (1998). Estimating root respiration, microbial respiration in the rhizosphere, and root-free soil respiration in forest soils. *Soil Biology and Biochemistry*, *30*(7), 961–968.

Landsberg, J. J., Gower, S. T. (1997). Applications of physiological ecology to forest management. San Diego, CA: Academic.

Lee, M. S., Nakane, K., Nakatsubo, T., Koizumi, H. (2003). Seasonal changes in the contribution of root respiration to total soil respiration in a cool-temperate deciduous forest. *Plant and Soil*, *255*(1), 311–318.

Melillo, J. M., McGuire, A. D., Kicklighter, D. W., Moore, B., III, Vorosmarty, C. J., Schloss, A. L. (1993). Global climate change and terrestrial net primary production. *Nature*, *363*(6425), 234–240.

Moreno, F. H. (2004). Soil carbon dynamics in primary and secondary tropical forests. Miami, FL: Florida International University.

Nemani, R. R., Keeling, C. D., Hashimoto, H., Jolly, W. M., Piper, S. C., Tucker, C. J., et al. (2003). Climate-driven increases in global terrestrial net primary production from 1982 to 1999. *Science*, *300*(5625), 1560–1563.

Nepstad, D. C., Uhl, C., Serrao, E. A. S. (1991). Recuperation of a degraded Amazonian landscape: forest recovery and agricultural restoration. *Ambio*, *20*(6), 248–255.

O'Brien, J. J., Oberbauer, F. S. (2001). An inexpensive, portable meter for measuring soil moisture. *Soil Science Society of America Journal*, *65*, 1081–1083.

Persson, H. (1990). Methods of studying root dynamics in relation to nutrient cycling. In A. F. Harrison, P. Ineson & O. W. Heal (Eds.), Nutrient cycling in terrestrial ecosystems. Field methods, application and interpretation (pp. 198–217). Elsevier Applied Science.

Phillips, O. L., Malhi, Y., Higuchi, N., Laurance, W. F., Nunez, P. V., Vasquez, R. M., et al. (1998). Changes in the carbon balance of tropical forests: evidence from long-term plots. *Science*, *282*(5388), 439–442.

Raich, J. W. (1983). Effects of forest conversion on the carbon budget of a tropical soil. *Biotropica*, *15*(3), 177–184.

Raich, J. W., Nadelhoffer, K. J. (1989). Belowground carbon allocation in forest ecosystems: global trends. *Ecology*, *70*(5), 1346–1354.

Raich, J. W., Schlesinger, W. H. (1992). The global carbon-dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus Series B-Chemical and Physical Meteorology*, *44*(2), 81–99.

Rodrigues, W., Furch, K., Klinge, H. (2001). Comparative study of the litterfall in a primary and secondary terra firme forest in the vicinity of Manaus, State of Amazonas, Brazil. *Amazoniana-Limnologia Et Oecologia Regionalis Systemae Fluminis Amazonas*, *16*(3–4), 441–462.

Saugier, B., Roy, J., Mooney, H. A. (2001). Estimations of global terrestrial productivity: converging toward a single number? In J. Roy, B. Saugier & H. A. Mooney (Eds.), Terrestrial global productivity (pp. 543–557). San Diego, CA: Academic.

Schimel, D. S., Braswell, B. H., Holland, E. A., McKeown, R., Ojima, D. S., Painter, T. H., et al. (1994). Climatic, edaphic, and biotic controls over storage and turnover of carbon in soils. *Global Biogeochemical Cycles*, *8*(3), 279–293.

Schlesinger, W. H. (1997). Biogeochemistry: an analysis of global change (2nd ed.). San Diego, CA: Academic.

Sierra, C. A., Harmon, M. E., Moreno, F. H., Orrego, S. A., Del Valle, J. I. (2007). Spatial and temporal variability of net ecosystem production in a tropical forest: testing the hypothesis of a significant carbon sink. *Global Change Biology*, *13*(4), 838–853.

Silver, W. L., Ostertag, R., Lugo, A. E. (2000). The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. *Restoration Ecology*, *8*(4), 394–407.

Vogt, K. A., Vogt, D. J., Bloomfield, J. (1998). Analysis of some direct and indirect methods for estimating root biomass and production of forests at an ecosystem level. *Plant and Soil*, *200*(1), 71–89.

Wuebbles, D. J., Jain, A., Edmonds, J., Harvey, D., Hayhoe, K. (1999). Global change: state of science. *Environmental Pollution*, *100*, 57–86.

Carbon Sequestration Potential of Mangroves in Southeast Asia

Mohd Nazip Suratman*

1 Introduction

Mangrove forests are considered to be a unique and complex major component of coastal zones in the tropical and sub-tropical regions. They represent transitional ecosystems where the ocean, land, and freshwater meet. Their main vegetation components are generally evergreen trees or shrubs that grow along coastlines, brackish estuaries or delta habitats. Mangrove habitats are easily recognized as they are located at tideland mud or sand flats inundated daily with sea water. They not only play critical roles in ensuring sustainability of coastal ecosystems, but also in fulfilling important socio-economic benefits to coastal communities.

Mangroves are formed in swampy areas with low oxygen conditions prevailing below the first few centimeters. Mangrove trees produce aerial roots that have the three functions of providing gas exchange, water uptake, and support to older trees. To overcome the low amount of oxygen, many mangrove trees including *Sonneratia* spp., *Avicennia* spp., *Brugueira* spp., C*eriops* spp., and *Rhizophora* spp. develop aerial root systems above the anaerobic substrate to allow gas exchange through the lenticels of the roots (Tomlinson, 1986). These roots also provide lateral support for the trees permitting them to withstand waves and winds in the environment (Lugo, 1990). Many mangrove species, such as those listed above, produce seeds that germinate and develop into seedlings while still attached to the parent tree known as vivipary (Duke, 1992; Abbott, 1994). Viviparity allows propagules to root more rapidly after dispersal when lodged in the mud. This is an important adaptation for propagules establishing themselves in tidal environment (Lugo, 1990).

Mangrove forests' latitudinal distributions are correlated with sea-surface temperature, and tend to mirror coral reef and sea grass bed conditions (Duke, 1992). Optimal growth conditions for mangroves are high humidity, high year-round

---

Faculty of Applied Sciences, Universiti Teknologi MARA (UiTM), 40450 Shah Alam, Malaysia,
*nazip@salam.uitm.edu.my

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

rainfall, a large intertidal range, and a moderate amount of land runoff (Clough, 1993). With many areas containing these conditions, Southeast Asia contains the richest species composition and more than one-third of the world's mangroves (FAO, 1985). Mangrove ecosystems may provide important nutrients and organic carbon to tropical coastal oceans (Alongi, 1996). In addition, they may intercept terrestrial-derived nutrients, pollutants and sediments before they reach the coastal ocean, alleviating problems due to high loading of these anthropogenic constituents (Valiela & Cole, 2002).

Similar to the tropical rainforests, mangrove forests continue to be degraded at rapid rates through different types of human activities (Saenger et al., 1983; Hamilton & Snedaker, 1984). In contrast to tropical rainforests, however, mangroves lack high level of research and conservation efforts, which result in limited understanding of these ecosystems. As mangrove ecosystems represent substantial connections between coastal habitats and terrestrial systems, their degradation could affect the ecological stability of coastal zones.

The purpose of this chapter is to provide an overview of the potential roles of mangroves in sequestering carbon from the atmosphere. The potential importance of mangroves in the global carbon cycle and in offsetting global climate change is highlighted. Specifically, this chapter describes the distribution and abundance of mangroves, discusses the major ecological and economical roles, and highlights the contribution of mangroves for conservation of biodiversity. In addition, the major threats to these ecosystems and their rates of destruction are also discussed. Finally, the potential of mangrove biomass to offset carbon emission in contributing carbon content stabilization in the atmosphere is discussed.

2 Distribution and Abundance

2.1 Global

While there are general estimates available on total area of mangrove forests in the world, the extent on the globe is not exactly known. This could be due to differences in assessment methods, definitions, or inconsistent sources of statistics in different countries. Table 1 presents a global extent of mangrove area estimates derived from various published sources.

The first attempt to provide global mangrove area estimates was undertaken in 1980 by FAO as part of Tropical Forest Resource Assessment initiatives (FAO, 1981). The total of mangrove areas in that year was estimated to be 15.6 million hectares, based on data sets collected from a total of 51 countries (Table 1). More recent attempts showed that the estimates of world mangroves are found to be vary, which could partly be due to the different numbers of countries included in the estimates.

Table 1 Global extent of previous mangrove area estimates (FAO, 2003)

Reference	Reference year[a]	Number of countries included	Estimated total area (ha)
FAO/UNEP (1981)	1980	51	15,642,673
Saenger et al. (1983)	1983	65	16,221,000
FAO (1994)	1980–1985	56	16,500,000
Groombridge (1992)	1992	87	19,847,861
ITTO/ISME[b] (1993)	1993	54	12,429,115
Fisher and Spalding (1993)	1993	91	19,881,800
Spalding et al. (1997)	1997	112	18,100,077
Aizpuru et al. (2000)	2000	112[c]	17,075,600
FAO (2003)	2003	112	14,653,000

[a] Except for FAO/UNEP (1981) and Aizpuru et al. (2000), the reference year is the year of the publications in which the estimate is cited, not the weighted average of all the national area estimates

[b] Combined figure from three publications by Clough (1993), Diop (1993), and Lacerda (1993)

[c] New estimates were provided for 21 countries, and for the remaining countries the study relied on Spalding et al. (1997)

The most recent estimates by FAO, taking into account a total of 112 countries, indicated that the current mangrove area is 14.6 million hectares world wide (FAO, 2003). This figure suggests that the world extent of mangrove areas decreased about 1 million hectare or 6.4% in two decades. This estimate was undertaken based on a collection of more than 2,800 national and sub-national data sets, that covered all included countries and areas where mangroves are known to exist, with the earliest estimates dating back to 1918. This is followed by a compilation of an updated list of the most recent, reliable estimate for each country. Regression analyses were performed using existing reliable data over time for each country, which provided estimates for 1980 and 1990 and an extrapolated estimate for 2000 for each country.

2.2 Southeast Asia

Southeast Asia covers an area of about 4,100,000 km^2. It is a sub-region of Asia, consisting of eleven countries that are geographically located at south of China, east of India and north of Australia. This sub-region consists of two geographic regions: the mainland and maritime. The mainland consists of Cambodia, Laos, Myanmar, Thailand and Vietnam whereas the maritime section consists of the Brunei, Timor Leste, Indonesia, Malaysia, Philippines and Singapore.

Southeast Asian mangroves recorded the highest diversity of all the world's mangroves (Fig. 1). An estimate of the spatial extent of mangroves in Southeast

Asia reported that the region contains over 6.1 million hectares of mangroves, approximately 35% area of global mangrove vegetation. In terms of species diversity, Southeast Asia holds nearly 75% of the world's mangrove species with the highest species diversity found in Indonesia with 45 species, followed by Malaysia (36 species) and Thailand (35 species) (Table 2). A list of plants commonly found in mangrove forests in Malaysia is presented in Appendix.

The global distribution of mangroves stretches into the higher latitudes to about 40° north and south with the main distribution in the tropical and sub-tropical areas, and the majority of mangrove areas in Southeast Asia (Ong et al., 1995; FAO, 2003). The largest area of mangrove occurs in the zone from 0° to 10° of both north and south latitudes (Twilley et al., 1992). As can be seen in Table 2, the most extensive area of mangroves in Southeast Asia found is Indonesia, followed by Malaysia and Myanmar. These three countries account for about 86% of all mangroves in Southeast Asia.

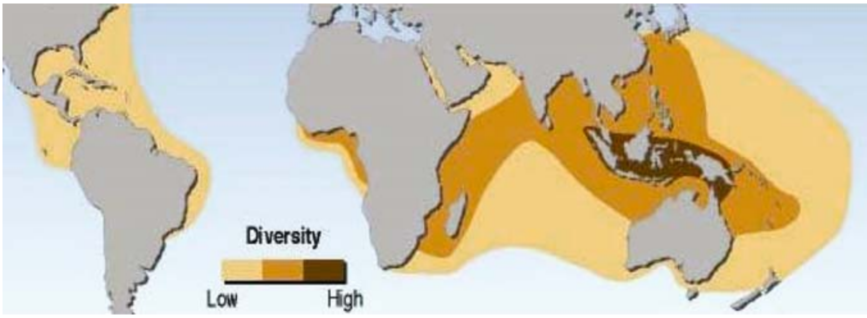


Fig. 1 Global distribution of mangrove diversity (UNEP, 2002)

Table 2 Mangrove area estimates for Southeast Asia (FAO, 2003)

Country	Mangrove area (ha)	Reference year	No. of mangrove species
Brunei	17,100	1992	29
Cambodia	72,835	1997	5
Indonesia	3,493,110	1988	45
Laos	N/A	N/A	N/A
Malaysia	587,269	1995	36
Myanmar	452,492	1996	24
Philippines	127,610	1990	30
Singapore	500	1990	31
Thailand	244,085	2000	35
Timor Leste	3,035	2000	N/A
Vietnam	252,500	1983	29

3 Mangrove Forest Destruction

Despite a better understanding towards their diversity and multiple roles, mangrove ecosystems are identified as among the most threatened habitats on Earth. Mangrove habitat destruction through human encroachment has been the primary cause of mangrove loss. Findings from mangrove resource assessments revealed alarming levels of mangrove destruction. Although scientists cannot determine exactly how extensive mangroves were before people began to alter coastlines, based on historical records, anywhere from 5% to nearly 85% of original mangrove areas in various countries is believed to have lost (World Resources Institute, 2000). Over the last 50 years or so, mangroves have disappeared and continued to be degraded by many activities such as land reclamation, sedimentation, pollution, extraction, conversion, prolonged flooding, and freshwater intrusion.

Tabuchi (2003) estimated that on a global scale, the area under mangroves is shrinking by 100,000 hectares annually due to clear cutting of timber and conversion into aquaculture projects. The loss of mangroves in Gili Petagan, Indonesia occurred as a result of intensive extraction of fuel-wood during 1970s. In Thailand, about 65% of the area of mangroves was lost during 1976–1991 due to an expansion of shrimp farming in the central, east coast, and south east regions of the country (Aksornkoae et al., 1993); MacKinnon (1997) estimated that by the early 1990s, Myanmar had lost almost 75% of the original extent, Vietnam had lost 37%, and Thailand 84%. By the mid of 1980s, Brunei and Philippines had lost 20% and 67% of their original mangrove cover, respectively. Earlier estimates suggested that by early 1980s, Indonesia had lost 55% of its mangroves. Chan et al. (1993) estimated that Malaysia had lost 12% of its mangrove forests between 1980 and 1990. According McKinnon (1997), by 1993, about 74% of mangrove area in Malaysia was lost from their original extent. In a few regions (i.e., Latin America and the Caribbean), however, the mangrove area is increasing as a result of plantation forestry and natural regeneration. The lack of adequate data on changes in mangrove for some Southeast Asian countries prevents efforts to report trends for the region as a whole.

In the most recent study, Ahmad and Suratman (2007) conducted a change detection analysis of mangrove forests utilizing a time series of Landsat TM imagery in Pulau Indah (literally translated means “Beautiful Island”) and its vicinity, Malaysia. This analysis was focused on determining mangrove reduction rates and identifying their spatial patterns within two interval periods, 1995 to 1999 and 1999 to 2005 (Fig. 2). Results from the analysis suggested there has been a decline in mangrove forests during both intervals at the rates of 14.1% and 1.9%, respectively. The higher declining rate during the first interval was primarily due to expansion and land development for building seaport infrastructures on this island. The estimated reduction rate of mangrove forests for the 10 year-period was 1.6% per year, which is slightly higher that reported nationally by Ong (1982) over the past 20 years (i.e., 1% per year).

Fig. 2 Decline of mangrove forests in Pulau Indah, Malaysia. Area in dark green represents mangrove forests (Ahmad & Suratman, 2007)

4 Services and Functions Provided by Mangroves

4.1 Resources and Socio-economic Contribution

In terms of area, mangrove forests only occupy about 8% of the world's coast (World Resources Institute, 2000). However, despite their small proportion, they play important roles in global ecosystem. In many developing nations in the tropics, mangrove forests have been traditionally utilized by the indigenous and local people for a variety of purposes. They have been dependent on mangrove waters for centuries for fishes, shrimps, crabs, and mollusks and will continue to rely on these products from mangrove habitats. It has been estimated that 60% of the commercial catch in Fiji depends on mangroves (Hamilton & Snedaker, 1984). Along the Carribean coast of Colombia, approximately 70% of the local fisheries are dependent on mangrove swamps in the Cienaga Grande (Bossi & Citron, 1990).

Mangroves also provide local people with a large variety of timber for construction, firewood, charcoal, poles, fishing gear, and etc. Other products collected from mangrove include fruit, honey, pulp, tannin, (Hamilton & Snedaker, 1984) and traditional medicines from many parts of the plants (Bandaranayake, 1998).

4.2 Ecological and Biodiversity Conservation

The important ecological functions and conservation of biodiversity of mangroves are being recognized in a number of studies. Among the most commonly known and discussed roles of mangroves are those of providing coastal protection against storms, waves and water currents, thus reducing erosion and flooding (Othman, 1994). Mangroves also perform an important function in impeding saltwater intrusion. The ability of mangrove forests to provide shoreline stabilization and storm protection are due to their location in hurricane and typhoon-prone tropical areas

(Primavera, 1995). Prop roots, deeply anchored in mud, help enhance soil consolidation and stability, which provides a good form of natural coastal protection (Siddiqi, 1993). A number of studies have confirmed the function of mangroves in terms of a green belt (Ruitenbeek, 1992; Janssen & Padilla, 1999; Suthawan, 1999; Danielsen et al., 2005; IUCN, 2005). According to Danielsen et al. (2005), an area of mangroves, 100 m wide, with a mangrove density of two or three trees every metre could decrease the height of waves by an estimated 70%. Although some young mangroves may be destroyed by natural disasters, some mature mangroves can survive due their complicated root systems (IUCN, 2005).

There have been several studies showing the beneficial role of mangrove natural features that can help reduce the damage caused by extreme natural phenomena such as tsunami. Many articles highlighted mangrove ecosystems, because they are one of the most dominant ecosystems in coastal areas, particularly those located in the tsunami-impacted regions. For example, according to Dahdouh-Guebas (2006), the areas covered with mangroves in the Andaman Islands received only 7% of damage, while the areas of the Islands where mangrove had been degraded had an estimated 80% to 100% of the damage. Ghosh (2005) reported that in the southern part of India, with dense mangroves, fewer people suffered from a tsunami and less property was destroyed compared to the areas without mangroves. In Malaysia, Emmanuel and John (2005) reported that Penang Island showed less effect from the waves since the presence of mangrove forests acted as riparian buffers.

The importance of mangrove ecosystems in conservation of biodiversity continues to be a topic of interest. Mangroves provide protective habitat for spawning, nursery, and feeding ground for juvenile fish and crustacean species that spend part of their lives in these habitats (Sasekumar et al., 1994). Mangroves contribute many different functional ways to fisheries. These include contributing nutrients to support an elaborate food web within the mangroves, exporting-derived nutrients offshore to enhance fisheries, or providing habitat to fauna for shelter and nursery grounds (Robertson et al., 1992; Clough, 1993; Marshall, 1994). Mangroves also provide shelter to a wide variety of mammals, amphibians and reptiles. Many other animals such as birds, marine and terrestrial mammals use or visit the mangroves for roosting, nesting, or feeding on a daily or seasonal basis (Hamilton & Snedaker, 1984).

The flora and fauna diversity in mangrove ecosystems provide scientific study and offer tourism opportunities. For example, in Malaysia, Kampung Kuantan is well known for its unique twinkling fireflies (*Photuris lucicrescens*) that live on *Sonneratia caseolaris* trees growing gregariously along river banks. Fireflies are found in marshes or in wet areas where their larvae have more abundant sources of food. The diet of adult fireflies feed on nectar from *S. caseolaris*.

Mangroves also play important roles in creating healthy coastal ecosystems. Adeel and Pomeroy (2002) reported that leaf litter from mangroves provides a principal source of nutrients for the trophic food as the ecosystems produce high productivity of organic matter. For example, the average of leaf litter input rate is 100 g m^{-2} $year^{-1}$ despite relatively low standing biomass accumulation averaged at 1,500 g m^{-2}. The high productivity is often attributed to high litter degradation rates and efficient recycling of nutrients (Bouillon et al., 2002).

5 Carbon Sequestration in Mangrove Forests

The essential elements of the carbon cycle are shown in schematic diagram of Fig. 3. The carbon budget is closely linked to three major components: terrestrials, coastal and oceans (Twilley et al., 1992). Coastal ecosystems are narrow transition zones between terrestrial and ocean areas, dominated by mangroves, shallow submerged coral reefs and sea grasses coastal oceans. The rate at which carbon cycles through the ecosystem is determined by a number of processes, particularly the rates of primary productivity and decomposition. Both processes are strongly influenced by environmental conditions such as temperature and precipitation. As part of the carbon cycle, autotrophs from terrestrial and coastal ecosystems acquire CO_2 in the atmosphere from photosynthesis by diffusion through leaf stomata, thereby incorporating the CO_2 into their biomass. Some of biomass becomes a carbon source for consumers and respiration returns CO_2 to the atmosphere. Photosynthesis and respiration form a link between terrestrial and coastal ecosystems and the atmosphere (Fig. 3). Carbon loss by photosynthesis is balanced by carbon release during respiration.

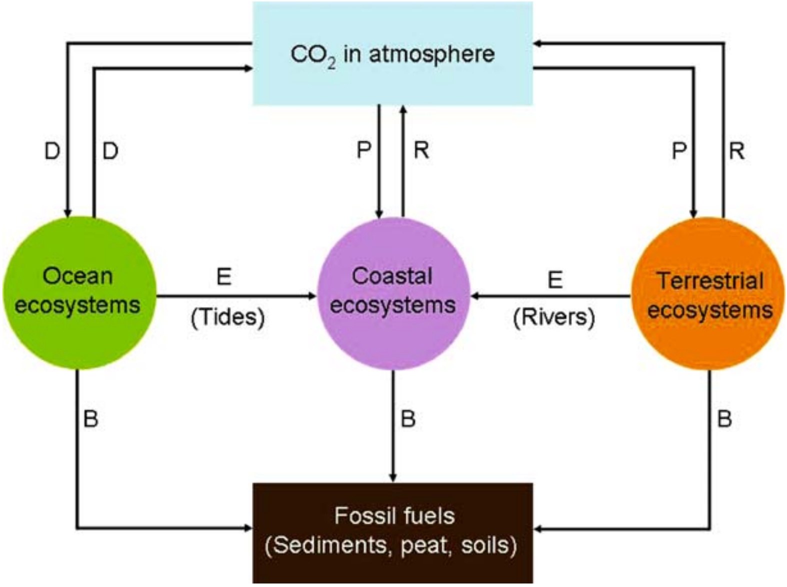


Fig. 3 A simple model for the carbon cycle as it occurs in ocean, coastal and terrestrial ecosystems. D = diffusion, P = photosynthesis, R = respiration, E = exchange, B = burial (Adapted from Twilley et al., 1992; Smith & Smith, 2003)

In ocean ecosystems, photosynthesis and respiration are also important but carbon cycling is more complex due to interaction of CO_2 with water. As the oceans contain more CO_2 than atmosphere and the land biosphere, CO_2 moves between the atmosphere and the ocean by molecular diffusion when there is a difference in CO_2 gas pressure between the atmosphere and oceans. If CO_2 concentrations in the atmosphere and the surface water are in equilibrium, the net exchange would stop. However, this does not seem to occur, because the concentration of CO_2 in the atmosphere continues to rises as a result of fossil fuel combustion (Smith & Smith, 2003).

Dissolved CO_2 reacts with water to form carbonic acid, which reacts with limestone to form bicarbonates and carbonate ions. As CO_2 is used in photosynthesis, bicarbonates are reconverted to $CO_{2.}$ Thus bicarbonates serve as a CO_2 reservoir and some aquatic autotrophs can use dissolved bicarbonates directly as a carbon source.

Being strategically located between land and sea, mangroves in the coastal ecosystems are important interfaces in the exchange of sediment, organic materials and gasses between coastal and both terrestrial and ocean ecosystems. In coastal ecosystems, where dead material from mangrove forests fall into the water, organic material does not completely decompose. Carbon is stored as formed fossil fuels, which are created by the burial of raw humus, partially decomposed organic matter and peat. This potential for storage of carbon in peat may be an important carbon sink and will be discussed later in this chapter.

Mangrove forests may play an important role in the carbon cycle in removing CO_2 from the atmosphere and storing it as carbon in plant materials and soils in a process called sequestration. As about half of mass in trees is carbon, large amounts of carbon are potentially stored in mangrove forests and they may be the largest stores of carbon in coastal zones. To better understand the dynamic of organic matter cycling in mangroves, it is important to know the productivity of mangrove forests mainly in terms of primary production, biomass, litter production, carbon accumulation, and burial.

5.1 Primary Production

Primary production refers to the amount of material produced. It occurs through photosynthesis whereby mangrove plants convert solar energy, carbon dioxide, and water to glucose and eventually plant tissue. In coastal wetlands, this process is affected by changes is the physical and chemical environments, including the amount of solar radiation, water, nutrients, temperature, tides, soil type, oxygen concentration and pH (Twilley et al., 1992). In a study of tree growth and productivity in mangrove forests in Matang, Malaysia, Putz and Chan (1986) monitored the growth of *Rhizophora apiculata* trees from 1920 through 1981. They found that the net primary productivity rate of *R. apiculata* averaged 177 g m^{-2} $year^{-1}$. The mean mortality rate for trees more than 10 cm dbh was 3% per year. In another study, Ong et al. (1979) reported that the net productivity rate of a managed mangrove forest in Malaysia ranged from 118 to 241 g m^{-2} $year^{-1}$. In a study of the same tree species in Thailand, Aksornkoae (1975) estimated that the total net production of trees 11 to 14 years old was 24 g m^{-2} $year^{-1}$. Christensen (1978) found that primary production rate for this species in mangrove forests of Phuket Thailand was 200 g m^{-2}

year^{-1}. In another study, Ong et al. (1995) estimated the biomass productivity rate in terms of above-ground (AB) net productivity for *R. apiculata* from Matang mangrove forest in Malaysia was 1,224 g m^{-2} year^{-1}.

5.2 Biomass

In many cases, estimates of mangrove biomass are usually restricted to AB structures. There are only a few studies of estimation of both AB and below-ground (BG) biomass productions have been reported in Southeast Asia. For example, Komiyama et al. (2000) estimated that the total of AB and BG biomass accumulation of *Ceriops tagal* in southern Thailand was 1,798 g m^{-2}. The components of biomass accumulation, including the stem, branch, leaf, prop root and BG root were estimated as 534, 236, 133, 20, and 875 g m^{-2}, respectively. BG biomass of *C. tagal* was found to be higher than AB biomass which gave an AB:BG ratio of 1:05. In contrast, Twilley et al. (1992) found that the AB was higher than that of BG biomass in their global estimate of mangrove biomass. The AB and BG accumulation were 1,782 and 1,463 g m^{-2}, respectively, which gave the AB:BG ratio of 0.82. In Malaysia, AB biomass accumulation for *R. apiculata* ranged from 2,700 to 4,600 g m^{-2} with a mean of 4,090 g m^{-2} (Putz & Chan, 1980).

According to Twilley et al. (1992), the distribution of biomass accumulation throughout the tropical regions indicated that higher values occur at lower latitudes. Biomass production of mangrove decreased with latitude and a model was fit to the data resulting in: $Y = -7.29(X) + 298.5$ ($R^2 = 0.75$), where X is latitude and Y is predicted biomass (g m^{-2}). This model was used to estimate the distribution of AB biomass of mangrove forests in latitudinal zones.

5.3 Litter Production and Decomposition

Mangroves are widely recognized as productive sources of plant debris which enters estuaries in the form of fallen leaves, twigs, and reproductive structures (May, 1999). Litter production is a commonly measured functional aspect of mangrove forests. It is an important component of the cycling of carbon and nutrients in mangrove ecosystems (Roderstein et al., 2005) and contributes to the food chains, through a leaf-detritus pathway, in the benthic coastal systems (Snedaker, 1978). The dissolved organic carbon (DOC) and particulate organic matter (POM) derived from this pathway becomes an important contributor to estuarine secondary productivity (Twilley, 1982); Snedaker (1978) reviewed important aspects of mangrove ecosystems and suggested that "the value of mangroves is accepted and based largely on the production of organic matter as leaf litter fall that enters the estuarine system, where it forms the basis of a complex food web". Based on this information, global or local patterns of AB dynamics in mangrove ecosystems can be described.

The rate of litter decomposition can be affected by a number of factors, including species, tidal height, rainfall, feeding activities of marine invertebrates, temperature, which in turn affects the rates of organic carbon export from mangrove ecosystems (Twilley, 1982; Camilleri, 1992; Mackey & Smail, 1996; Wafar et al., 1997; Woitchik et al., 1997). As an example, litter decomposition rates vary between mangrove species. Kristensen et al. (1995) found that *Avicennia* spp. leaves decompose faster than those of *Sonneratia* spp. and *Rhizophora* spp. because they are thinner, contain lesser tannins, and sink faster. Lu and Lin (1990) found that litter of *Bruguiera* spp. decomposes quickly, in contrast, *Aegiceras* spp. decomposes slowly (Tam et al., 1990).

Twilley et al. (1992) reported that values of litter productivity from mangrove forests world-wide range from 20 to 160 g m^{-2} $year^{-1}$. In Tritih, central Java, Indonesia, Sukardjo and Yamada (1992) estimated that the litter production rate of *R. mucronata* plantations ranged from 71 to 104 g m^{-2} $year^{-1}$ of dry weight with leaf litter making up 73–84% of the total. Litter fall, leaf production, and propagule production were measured by Clough et al. (2000) in monocultures of the mangrove *R. apiculata* aged from 6 to 36 years in Ca Mau Province, southern Vietnam. They found that total annual litter fall ranged from 94 to 188 g m^{-2} $year^{-1}$ of dry weight. They observed a similar trend as Sukardjo and Yamada's where leaf litter was the main component of litter fall in all stands studied.

The variability of export of leaf litter and detritus from mangrove zones has been attributed to the differences in hydrology, mangrove types, dominant species and stages of development (Odum et al., 1982; Day et al., 1996; Chen & Twilley, 1999). Twilley et al. (1992) estimated that export of organic carbon from mangrove forests ranges from 10% to 50% of litter production. In Malaysia, Gong and Ong (1990) determined the quantitative relationship between the export of material and the areal extent of mangroves in Matang mangrove forest reserve. Using the figure of 50% export, the export of biomass through leaf litter was estimated as 193.5 g C m^{-2} $year^{-1}$. This information supports the importance of mangroves in the export of organic carbon, and also their role as carbon sinks.

Many studies have estimated the total carbon and nutrient accumulation in mangrove forests but limited studies have been conducted for Southeast Asia. BG carbon accumulation was examined for mangrove forests in Pohnpei Island, Micronesia by Fujimoto et al. (1999). They estimated that carbon accumulation rate was 93 g C m^{-2} $year^{-1}$ in conjunction with a sedimentation rate of 2 mm $year^{-1}$ during sea-level rise phases, but may be higher in other sites where higher accretion rates have been observed. In western Australia, Alongi et al. (2003) estimated that the BG (in soil, live and dead roots) carbon accumulation for *R. stylosa* and *Avicennia marina* forests ranged between 1,400 to 3,300 g C m^{-2} and 1,200 to 3,600 g m^{-2}, respectively.

In subtropical mangrove forests, senescent leaves of *R. mucronata* and *C. tagal* decomposed substantially faster during rainy season (Woitchik et al., 1997). Mackey and Smail (1996) found significantly faster decomposition of *A. marina* debris in lower intertidal zones with greater inundation. They also found an exponential relationship between leaf decomposition rate and latitude. Leaves decomposed more rapidly at lower latitudes. They attributed the pattern to temperature differences, and concluded that seasonality may have important effects on organic cycling and nutrient export from mangrove ecosystems.

According to Camilleri (1992), feeding activities of marine invertebrates have important effects on decomposition rates of mangrove litter. Marine invertebrates may process large volume of the litter, therefore contributing to nutrient dynamics. For example, in Ao Nam Bor mangrove forest in Thailand, the analysis of foregut contents of sesarmid crabs indicated that their diets contained 55–82% of mangrove plant matter (Poovachiranon & Tantichodok, 1991).

The function of mangroves as carbon storage ecosystems has been further reviewed based on published information on the accumulation of organic matter as peat. Mangrove peat has been reported to vary from 2 m to over 9 m depth (Fujimoto et al., 1999; Macintyre et al., 2004), which suggests large stores of plant biomass beneath some mangrove forests. Golley et al. (1962) examined the partition of the total biomass (AB and BG) contained in a mangrove tree and found that peat and fine rots (<0.5 cm diameter) exceeded all other biomass components combined by 5:1. This accumulation of organic material as peat in mangrove soils serves as a sink for carbon and other nutrients.

Mangrove forests are usually characterized by sediment accretion (Woodroffe, 1992). This fact, combined with the high productivity and low ratio of sediment respiration to net primary production, gives mangrove sediments the potential for long-term sequestration of organic carbon. Thus, these systems may play an important role in global carbon cycle (Jennerjahn & Ittekkot, 2002).

Several methods have been used to measure sedimentation rates, including the use of natural and man-made radionuclides, such as ^{137}Cs and ^{210}Pb. Alongi et al. (2001) used these methods to estimate mass sediment accumulation in four mangrove forests in Ao Sawi, Thailand. The mass sediment accumulation rate was rapid at all forests studied, ranging from 2,900 to 7,600 g m^{-2} $year^{-1}$. Using the same method, a higher rate of mass sediment accumulation rate was recorded (2,200–11,400 g m^{-2} $year^{-1}$) for Matang mangrove forests in Peninsular Malaysia (Alongi et al., 2004). In Ba Lat estuary, Vietnam, sediment rates were compared between dry and wet seasons by Van Santen et al. (2007) using sediment traps in a bare mudflat containing a vegetation gradient from pioneering mangroves to 5- to 7-year-old mangroves. As expected, the sedimentation rate recorded in the wet season was higher than in the dry season (34,600 g m^{-2} $year^{-1}$ versus 29,400 g m^{-2} $year^{-1}$).

Alongi et al. (2001) also examined rates of carbon mineralization and burial in sediments accumulation in Ao Sawi, Thailand and found that the thickness of mixed layer sediments in all forests ranged from 16 to 38 cm. Total inputs of organic carbon ranged from 26.4 to 40.9 mol C m^{-2} $year^{-1}$ and burial rates ranged from 15.3 to 23.4 mol C m^{-2} $year^{-1}$. Total rates of carbon mineralization ranged from 7.0 to 16.4 mol C m^{-2} $year^{-1}$.

Chmura et al. (2003) studied the component of the terrestrial biological carbon pool by compiling data sets for 154 sites in mangroves and salt marshes from the western and eastern Atlantic and Pacific coasts, as well as the Indian Ocean, the

Mediterranean Ocean, and the Gulf of Mexico. They found that the average carbon density of mangroves (0.055 g cm^{-3}) is significantly higher than salt marshes (0.039 g cm^{-3}), and the carbon accumulation rate for both wetlands is about 210 g m^{-2} $year^{-1}$. Considering global estimates of mangrove coverages, they calculated that carbon sequestration in mangroves is about 38 Tg C $year^{-1}$, which suggests that mangroves sequester carbon faster than terrestrial forests. Also, they suggested that the role of mangrove as carbon sinks may be ecologically important since decomposition in mangrove soils occurs mainly though sulphate reduction, which would contribute less to greenhouse gases.

5.4 Methane Emissions from Mangroves

One of the important greenhouse gases that contribute to a rise in global mean surface temperature is methane (CH_4). Wuebbles and Hayhoe (2002) reported that the current globally-averaged mixing ratio of methane concentration is 1,750 parts per billion, which is more than double that of pre-industrial times. The concern over the increasing rate of methane concentration has stimulated the effort to develop policies for controlling its emission.

Methane emissions from mangroves show large variability due to the complexity of environmental factors that affect the production of methane. These include salinity, soil texture, temperature and nutrient, transport of organic sediments, vegetation types and morphology, chemical characteristics of organic materials, topography and geomorphology (Dunfield et al., 1992; Lindau et al., 1993). The supply of organic materials is important for methane production and emission (Wang et al., 1987); Bartlett et al. (1983) reported that salinity and sulphate are the major inhibitors of methane production by stimulating activity of sulphate reducing bacteria, which compete with methanogens for the reduced substrates. Soil texture is involved with transferring and trapping methane produced in the reduced soil and thus affects the net production of methane emission (Le Mer & Roger, 2001). According to Dunfield et al. (1993), soil temperature is critical in determining the production and emission of methane from the subsurface to the atmosphere with optimum temperature between 25°C to 30°C.

There are only a few investigations on methane emissions from mangroves in Asia (e.g., Alongi et al., 2001; Purvaja & Ramesh, 2001; Lekphet et al., 2005; Krupadam et al., 2007). Methane emissions from various mangrove areas have estimated with a wide range of emissions. Some studies found that methane emissions from mangrove were considered to be negligible (Giani et al., 1996; Alongi et al., 2001). In South India, Purvaja and Ramesh (2001) estimated the methane emission rate from mangrove forests ranging from 0.0473 to 0.3245 g m^{-2} day^{-1}. In Ranong Province, Thailand, Lekphet et al. (2005) reported that emission rates were found to vary seasonally with the highest rate in the rainy season, followed by summer and cold seasons, during which the values were 0.00052, 0.00027, and 0.00019 g m^{-2} day^{-1}, respectively. They reported that these seasonal variations were as a result from several factors in field conditions such as water conductivity, soil temperature, and water level. In southeastern coast of Puerto Rico, Sotomayor et al. (1994) estimated the methane emission rate from mangroves ranging from 0.0004–0.0082 g m^{-2} day^{-1}. Lyimo et al. (2002) found that the methane emission rate in Mtoni mangrove sediments Tanzania were at the range of 0–0.0192 g m^{-2} day^{-1}.

6 Conclusions

The main objective of this chapter was to review the roles of mangrove primary production in the carbon cycle of mangroves and in the coastal ecosystems, which continues to be an issue of interests. Studies have shown that mangrove ecosystems contribute to primary productivity of the coastal ecosystems and indirectly increasing secondary productivity of adjacent habitats. Published data compiled from differing mangrove ecosystems shows that organic carbon in mangroves and marine sediments contribute immensely in providing source of carbon. This information supports the importance of mangroves in the cycling of nutrients, and also their role in carbon sequestration.

Degradation of mangrove ecosystems through a variety of human activities has been the primary cause of mangrove loss. These impacts are likely to continue as human populations and land development activities expand further into mangrove ecosystems. Therefore, reducing mangrove degradation, reforestation of mangroves, and sustainably managed existing mangroves, can provide a cost-effective way of increasing their capacity in the carbon and nutrient cycles. However, to accurately calculate how much carbon can be stored and how to maintain them as sinks will be critical for those making decisions related to mangrove rehabilitation and reforestation. Hence, a better understanding on how mangrove plants and soils absorb, store, and release carbon, also how these activities will affect the carbon cycle, is essential in order to better understand of how mangrove ecosystems function, especially in their role in stabilizing carbon content in the atmosphere.

References

Abbott, J.G. (1994). *Mangrove ecosystems: Prospects for rehabilitation in the context of biology, economics and politics*. Presented at Coastal Zone Canada Conference, September, Halifax, Canada.

Adeel, Z., Pomeroy, T. (2002). Assessment and management of mangrove ecosystems in developing countries. *Trees*, *16*, 235–238.

Ahmad, S., Suratman, M.N. (2007). *Detection of changes in mangrove forests using Landsat TM in Pulau Indah, Malaysia*. Paper to be presented at the Conference of Forestry and Forest Products (CFFPR 2007), 27–29 November 2007, Kuala Lumpur, Malaysia.

Aizpuru, M., Achard, F., Blasco, F. (2000). *Global assessment of cover change of the mangrove forests using satellite imagery at medium to high resolution*. EEC Research Project No. 15017-1999-05 FIED ISP FR – Joint Research Centre, Ispra.

Aksornkoae, S. (1975). *Structure, regeneration and productivity of mangroves in Thailand*. Doctoral dissertation, Michigan State University, MI, 109 p.

Aksornkoae, S., Paphavasit, N., Wattayakorn, G. (1993). Mangroves of Thailand: Present status of conservation, use and management, p. 133–183. In: Clough, B. (Ed.), *The economic and environmental values of mangrove forests and their present state of conservation in the south-east Asia/Pacific region*. Mangrove Ecosystems Technical Reports. International Society for Mangrove Ecosystems, Okinawa, Japan.

Alongi, D.M. (1996). The dynamics of benthic nutrient pools and fluxes in tropical mangrove forests. *Journal of Marine Research*, *54*, 123–148.

Alongi, D.M., Wattayakorn, G., Pfitzner, J., Tirendi, F., Zagorskis, I., Brunskill, G.J., Davidson, A., Clough, B.F. (2001). Organic carbon accumulation and metabolic pathways in soil of mangrove forests in Southern Thailand. *Marine Geology*, *179*, 85–103.

Alongi, D.M., Tirendi, F., Clough, B.F. (2003). Below-ground decomposition of organic matter in forests of the mangroves *Rhizophora stylosa* and *Avicennia marina* along the arid coast of Western Australia. *Aquatic Botany*, *68*, 2, 97–122.

Alongi, D.M., Sasekumar, A., Chong, V.C., Pfitzner, J., Trott, L.A., Tirendi, F., Dixon, P., Brunskill, G.J. (2004). Sediment accumulation and organic material flux in a managed mangrove ecosystem: Estimates of land-ocean-atmosphere exchange in Peninsular Malaysia. *Marine Geology*, *208*, 383–402.

Ashton, E.C., Macintosh, D. (2002). Preliminary assessment of the plant diversity and community ecology of the Semantan mangrove forest, Sarawak, Malaysia. *Forest Ecology and Management*, *166*, 111–129.

Bandaranayake, W.M. (1998). Traditional and medicinal uses of mangroves. *Mangroves and Salt Marshes*, *2*, 133–148.

Bouillon, S., Koedam, M., Raman, A.V., Dehairs, F. (2002). Primary producers sustaining macro-invertebrate communities in intertidal mangrove forests. *Oecologia*, *130*, 441–448.

Camilleri, J.C. (1992). Leaf litter processing by invertebrates in a mangrove forest in Queensland. *Marine Biology*, *114*, 1, 139–146.

Chan, H.T., Ong, J.E., Gong, W.K., Sasekumar, A. (1993). *The socio-economic, ecological and environmental values of mangrove ecosystems in Malaysia and their present state of conservation. The Economic and environmental values of mangrove forests and their present state of conservation in the South-East Asia/Pacific Region*, pp. 41–81. Mangrove Ecosystems Technical Reports Vol. 3.

Chen, R., Twilley, R.R. (1999). A simulation model of organic matter and nutrient accumulation in mangrove wetland soils. *Biogeochemistry*, *44*, 93–118.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, 17, 1111–1120.

Christensen, B. (1978). Biomass and primary production of *Rhizophora apiculata* in a mangrove in Southern Thailand. *Aquatic Botany*, *4*, 43–52.

Clough, B.F. (1993). *The economic and environmental values of mangrove forests and their present state of conservation in the South-East Asia/Pacific Region*. Mangrove Ecosystems Technical Reports Vol.3. ITTO/ISME/JIAM Project PD71/89. Rev. 1(F) Okinawa, Japan, ISME, 202 pp.

Clough, B.F., Tan, D.T., Phuong, D.X., Buu, D.C. (2000). Canopy leaf area index and litter fall in stands of the mangrove *Rhizophora apiculata* of different age in the Mekong Delta, Vietnam. *Aquatic Botany*, *66*, 311–320.

Danielsen, F., Sørensen, M.K., Olwig, M.F., Selvam, V., Paris F., Burgess, N.D., Hiraishi, T., Karunagaran, V.M., Rasmussen, M.S., Hansen, L.B., Quarto., A., Suryadiputra, N. (2005). The Asian tsunami: A protective role for coastal vegetation. *Science*, *310*, 5748, 643.

Day, J.W., Coronado-Molina, C., Vera-Herrera, F.R., Twilley, R., Rivera-Monroy, V.H., Alvarez-Guillen, H., Day, R., Conner, W. (1996). A 7 year record of aboveground net primary production in a southeastern Mexican mangrove forest. *Aquatic Botany*, *55*, 39–60.

Diop, E.S. (1993). *Conservation and sustainable utilization of mangrove forests in Latin America and Africa regions*. Part II - Africa, pp. 245–261. Mangrove Ecosystems Technical Reports Vol. 3. ITTO/ISME Project PD114/90. Okinawa, Japan, ISME, 262 pp.

Duke, N.C. (1992). Mangrove floristics and biogeography. In: Robertson, A.I., Alongi, D.M. (Eds.), *Tropical mangrove ecosystems*. American Geophysical Union Press, Washington, DC, pp. 63–100.

Dunfield, P., Knowles, R., Dumont, R., Moore, T.R. (1993). Methane production and consumption in temperate and sub artic peat soils: Response to temperature and pH. *Soil Biology and Biochemistry*, *25*, 321–326.

Emmanuel, M., John, E. (2005). *Counting the cost. The New Straits Times*. 9 January 2005.

FAO (1985). *Mangrove management in Thailand, Malaysia and Indonesia*. FAO Environment Paper, Rome, pp. 60.

FAO (1994). *Mangrove forest management guidelines*. FAO Forestry Paper 117, Rome, 319 pp.

FAO (2003). FAO's database on mangrove area estimates. By Wilkie, M.L., Fortuna, S. and Souksavat, O. Forest Resources Assessment Working Paper No. 62. Forest Resources Division, FAO, Rome (Unpublished).

FAO/UNEP (1981). *Tropical forest resources assessment project, forest resources of Tropical Asia*. FAO/UNEP, Rome, 475 pp.

Fisher, P., Spalding, M.D. (1993). *Protected areas with mangrove habitat*. Draft Report World Conservation Centre, Cambridge, 60 pp.

Ghosh, N. (2005). *Singapore not safe from tsunami. The New Straits Times*, 15 October 2005.

Giani, L., Bashan, Y., Holguin, G., Strangmann, A. (1996). Characteristics and methanogenesis of the Balandra lagoon mangrove soils Baja California Sur, Mexico. *Geoderma 72*, 146–160.

Gong, W.K., Ong, J.E. (1990). Plant biomass and nutrient flux in a managed mangrove forest in Malaysia. *Estuarine, Coastal and Shelf Science*, *31*, 519–530.

Groombridge, B. (1992). *Global biodiversity: Status of the earth's living resources*. WCMC/The National History Museum/IUCN/UNEP/WWF/WRI. Chapman & Hall, London, 594 pp.

Fujimoto, K., Imaya, A., Tabuchi, R., Kuramoto, S., Utsugi, H., Murofushi, T. (1999). Belowground carbon storage of Micronesian mangrove forests. *Ecological Research*, *14*, 409–413.

Golley, F.B., Odum, H.T., Wilson, W.F. (1962). The structure and metabolism of a Puerto Rico mangrove forest in May. *Ecology*, *43*, 9–19.

Hamilton, L.S., Snedaker, S.C. (Eds.) (1984). *Handbook for mangrove area management*. IUCN/UNESCO/UNEP. East-West Centre, Honolulu, HI.

ITTO/ISME (1993). *Project PD71/89. Rev. 1 (F)*. International Society for Mangrove Ecosystems (ISME), Okinawa, Japan.

IUCN (2005). *Early observations of tsunami effects on mangroves and coastal forests*. The World Conservation Union (IUCN). Retrieved 5 June 2007, from: http://www.iucn.org/info_and_news/press/water-wetlands-statement.pdf.

Janssen, R., Padilla, J.E. (1999). Preservation or conversion: Valuation and evaluation of a mangrove forest in the Philippines. *Environmental and Resource Economics*, *14*, 3, 297–331.

Jennerjahn, T.C., Ittekkot, V. (2002) Relevance of mangroves for the production and deposition of organic matter along tropical continental margins. *Naturwissenschaften*, *89*, 23–30.

Komiyama, A., Havanond S., Srisawatt, W., Mochida, Y., Fujimoto, K., Ohnishi, T., Ishihara, S., Miyagi, T. (2000). Top/root biomass ratio of a secondary *Ceriops tagal* (Perr) C.B. Rob. Forest. *Forest Ecology and Management*, *139*, 127–134.

Kristensen, E., Holmer, M., Banta, G.T., Jensen, M.H., Hansen, K. (1995). Carbon, nitrogen and sulfur cycling in sediments of the Au Nam Bor mangrove forests, Phuket, Thailand: A review. *Phuket Marine Biological Centre Research Bulletin*, *60*, 37–64.

Krupadam, R.J., Ahuja, R., Wate, R.W., Anjaneyulu, Y. (2007). Forest bound estuaries are higher methane emitters than paddy fields: A case study of Godavari estuary, East Coast of India. *Atmospheric Environment*, *41*, 4819–4827.

Lacerda, L.D. (1993). *Conservation and sustainable utilization of mangrove forests in Latin America and Africa regions*. Vol. 2, Part I - Latin America. Mangrove Ecosystems technical reports ITTO/ISME Project PD114/90 (F). Okinawa, Japan, 272 pp.

Lekphet, S., Nitisoravut, S., Adsavakulchai, S. (2005). Estimating methane emissions from mangrove area in Ranong Province, Thailand. *Songklanakarin Journal of Science and Technology*, *27*, 1, 153–163.

Le Mer, J., Roger, P.A. (2001). Production, oxidation, emission and consumption of methane by soils: A review. *European Journal of Soil Biology*, *37*, 25–50.

Lindau, C.W., Bollich, P.K., DeLaune, R.D., Mosier, A.R., Bronton, K.F. (1993). Methane mitigation in flooded Louisiana rice fields. *Biology and Fertility of Soils*, *15*, 174–178.

Lu, C., Lin, P. (1990). Studies on litterfall and decomposition of *Bruguiera sexangula* (Lour.) Poir, community on Hainan Island. *China Bulletin of Marine Science 47*, 1, 139–148.

Lugo, A.E. (1990). *Fringe wetlands*. In: Lugo, A.E., Brinson, M., Brown, S. (Eds.), *Forested wetlands: Ecosystems of the world 15*. Elsevier, Amsterdam, The Netherlands, pp. 143–169.

Lyimo, T.J., Pol, A., Camp, H.J.M. (2002). Methane emission, sulphide concentration and redox potential profiles in Mtoni mangrove sediment, Tanzania. *Western Indian Ocean Journal of Marine Science*, *1*, 1, 71–80.

Mackey, A.P., Smail, G. (1996). Spatial and temporal variation in litter fall of *Avicennia* marina (Forssk) Vierh. in the Brisbane River, Queensland, Australia. *Aquatic Botany*, 133–145.

Mackinnon, J. (Ed.) (1997). *Protected areas systems review of the Indo-Malayan realm*. Asian Bureau for Conservation, Canterbury.

Macintyre, I.G., Toscano, M.A., Lighty, R.G., Bond, G.B. (2004). Holocene history of the mangrove islands of Twin Cays, Belize, Central America. *Atoll Research Bulletin*, 510, 1–16.

Marshall, N. (1994). Mangrove conservation in relation to overall environmental considerations. *Hydrobiologia*, *285*, 303–309.

MOSTI (2003). *Wetlands: Conservation and wise use*. Ministry of Science, Technology and the Environment, Malaysia, 60 pp.

Odum, W.E., McIvor, C.C., Smith, T.J. (1982). *The ecology of the mangroves of south Florida: A community profile*. FWS/OBS-81/24. United States Fish and Wildlife Service, Office of Biological Services, Washington, DC.

Ong, J.E., Gong, W.K., Wong, C.H., Dhanarajan (1979). *Productivity of a managed mangrove forest in West Malaysia*. Paper presented at International Conference on Trends in Applied Biology in S.E. Asia. Universiti Sains Malaysia, Penang, Malaysia.

Ong, J.E., Gong, W.K., Clough, B.F. (1995). Structure and productivity of a 20-year-old stand of *Rhizophora apiculata* mangrove forest. *Journal of Biogeography*, *22*, 417–424.

Othman, M.A. (1994). Value of mangroves in coastal protection. *Hydrobiologia*, *285*, 277–282.

Poovachiranon, S., Tantichodok, P. (1991). *The role of sesarmid crabs in the mineralization of leaf litter of Rhizophora apiculata in a mangrove, Southern Thailand*. Research Bulletin of Phuket Marine Biological Centre, *56*, 63–74.

Primavera, J.H. (1995). Mangroves and brackishwater pond culture in the Philippines. *Hydrobiologia*, *295*, 303–309.

Purvaja, G.R., Ramesh, R. (2001). Natural and anthropogenic methane emission from coastal wetlands of South India. *Environment Management*, *27*, 547–557.

Putz, F.E., Chan, H.T. (1986). Tree growth, dynamics, and productivity in a mature mangrove forest in Malaysia. *Forest Ecology and Management*, *17*, 211–230.

Robertson, A.I., Alongi, D.M., Boto, K.G. (1992). Food chains and carbon fluxes. In: Robertson, A.I., Alongi, D.M. (Eds.), *Tropical mangrove ecosystems*. American Geophysical Union, Washington, DC, pp. 293–326.

Roderstein, M., Hertel, D., Leuschner, C. (2005). Above-and below-ground litter production in three tropical montane forests in southern Ecuador. *Journal of Tropical Ecology*, *21*, 483–492.

Ruitenbeek, H.J. (1992). *Mangrove management: An economic analysis of management options with a focus on Bintuni Bay, Irian Jaya*. EMDI Environmental Report 8, Environmental Management Development in Indonesia Project.

Saenger, P., Hegerl E.J., Davie, J.D.S. (1983). *Global status of mangrove ecosystems*. Commission on Ecology Papers No. 3. IUCN, Gland, Switzerland, 88 pp.

Sasekumar, A., Chong, V.E., Singh, H. (1994). The physical and chemical characteristics of the Matang mangrove waters. *Proceedings of the 3rd ASEAN-Australia Symposium on Living Coastal Resources*, *2*, 446–453.

Siddiqi, N.A. (1993). *The importance of mangroves to people in the coastal areas of Bangladesh*. In: Proceedings of the VII Pacific Science Inter-Congress Mangrove Session. Okinawa, 1 July 1993, pp. 5–10.

Smith, R.L., Smith, T.M. (2003). *Elements of ecology*, 6th Edition. Benjamin Cummings, Belmont, CA, 682 pp.

Snedaker, S.C. (1978). Mangroves: Their value and perpetuation. *Nature and Resources*, *14*, 3, 6–13.

Sotomayor, D., Corredor, J.E., Morell, J.M. (1994). Methane and emission from mangrove soil along the southeastern coast of Puerto Rico. *Estuaries*, *17*, 140–147.

Spalding, M.D., Blasco, F., Field, C.D. (Eds.) (1997). *World mangrove atlas*. The International Society for Mangrove Ecosystems, Okinawa, Japan, 178 pp.

Sukardjo, S., Yamada, I. (1992). Biomass and productivity of a *Rhizophora mucronata* Lamarck plantation in Tritith, Central Java, Indonesia. *Forest Ecology and Management*, *49*, 3–4, 195–209.

Suthawan, S. (1999). *Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: Case study of Surat Thani, South of Thailand*. EEPSEA (Economy and Environment Program for Southeast Asia) Research Report.

Tabuchi, R. (2003). *The rehabilitation of mangroves in Southeast Asia*. Retrieved 1 June 2007, from http://www.unu.edu/env/plec/marginal/proceedings/TabuchiCH12.pdf.

Tam, N.F.Y., Vrijmoed, L.L.P., Wong, Y.S. (1990). Nutrient dynamics associated with leaf decomposition in a small subtropical mangrove community in Hongkong. *Bulletin of Marine Science* *47*, 1, 68–78.

Tomlinson, P.B. (1986). *The botany of mangroves*. Cambridge University Press, Cambridge, 419 pp.

Twilley, R.R. (1982). *Litter dynamics and organic exchange in black mangrove* (*Avicennia germinans*) *basin forest in southwest Florida estuary*. Doctoral dissertation, University of Florida, Gainesville, FL.

Twilley, R.R., Chen, R.H., Hargis, T. (1992). Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. *Water, Air and Soil Pollution*, *64*, 265–288.

UNEP (2002). *Global distribution of coral, mangrove and sea grass diversity*. Retrieved 26 May 2007, from: http://maps.grida.no/go/graphic/distribution_of_coral_mangrov e_and_seagrass_diversity.

Valiela, I., Brown, J.L., York, J.K. (2001). Mangrove forests: One of the world's threatened major tropical environments. *BioScience*, *51*, 10, 807–815.

Van Santen, P., Augustinus, P.G.E.F., Janssen-Stelder, B.M., Quartel, S., Tri, N.H. (2007). Sedimentation in an estuarine mangrove system. *Journal of Asian Earth Sciences*, *29*, 4, 566–575.

Wafar, S., Untawale, A.G., Wafar, M. (1997). Litter fall and energy flux in a mangrove ecosystem. Estuarine Coastal and Shelf. *Science*, *44*, 111–124.

Wang, Z.P., Delaune, R.D., Patrick, W.H., Jr., Masscheleyn, P.H. (1993). Soil redox and pH effects on methane production in a flooded rice soil. *Soil Science Society of America Journal*, *57*, 386–391.

Woitchik, A.F., Ohawa, B., Kazungu, J.M., Rao, R.G., Goeyens, L., Dehairs, F. (1997). Nitrogen enrichment during decomposition of mangrove litter in an east African coastal lagoon (Kenya): Relative importance of biological nitrogen fixation. *Biogeochemistry*, *39*, 15–35.

Woodroffe, C.D. (1992). Mangrove sediments and geomorphology. In: Alongi, D., Robertson, A. (Eds.), *Tropical mangrove ecosystems*. American Geophysical Union, Coastal and Estuarine Studies, pp. 7–41.

World Resources Institute (2000). *World resources 2000–2001. People and ecosystems: The fraying web of life*. Elsevier Science, 389 pp.

Wuebbles, D.J., Hayhoe, K. (2002). Atmospheric methane and global change. *Earth-Science Review*, *57*, 117–210.

Appendix A list of plant species found in the mangrove forests in Malaysia (Saenger et al., 1983; Ashton & Macintosh, 2002; MOSTI, 2003)

No.	Family	Species	Category[a]	Life-form	Common name (in Malay)
1	Acanthaceae	*Acanthus ilicifolius*	MA	Shrub	Jeruju puteh
2	Arecaceae	*Nypa fruticans*	MA	Palm	Nipah
3	Asteraceae	*Pluchea indica*	MA	Shrub	Beluntas
4	Avicenniaceae	*Avicennia alba*	M	Tree	Api-api puteh
5	Avicenniaceae	*A. lanata*	M	Tree	Api-api bulu
6	Avicenniaceae	*A. marina*	M	Tree	Api-api jambu
7	Avicenniaceae	*A. officinalis*	M	Tree	Api-api ludat
8	Combretaceae	*Lumnitzera littorea*	M	Shrub/tree	Teruntum merah
9	Combretaceae	*L. racemosa*	M	Shrub/tree	Teruntum putih
10	Euphorbiaceae	*Excoecaria agallocha*	M	Tree	Buta-buta
11	Meliaceae	*Xylocarpus granatum*	M	Tree	Nyireh bunga
12	Meliaceae	*X. meluccensis*	M	Tree	Nyireh batu
13	Myrsinaceae	*Aegiceras corniculatun*	M	Shrub	Kachang-kachang
14	Myrsinaceae	*A. floridum*	M	Shrub	Kachang-kachang
15	Pteridaceace	*Acrostichum aureum*	M	Fern	Piai raya
16	Pteridaceace	*A. speciosum*	M	Fern	Piai lasa
17	Rhizophoraceae	*Bruguiera cylindrica*	M	Tree	Berus
18	Rhizophoraceae	*B. gymnorrhiza*	M	Tree	Tumu merah
19	Rhizophoraceae	*B. parviflora*	M	Tree	Lenggadai
20	Rhizophoraceae	*B. sexangula*	M	Tree	Tumu putih
21	Rhizophoraceae	*Ceriops decandra*	M	Tree	Tengar
22	Rhizophoraceae	*C. tagal*	M	Tree	Tengar
23	Rhizophoraceae	*Rhizophora apiculata*	M	Tree	Bakau minyak
24	Rhizophoraceae	*R. mucronata*	M	Tree	Bakau kurap
25	Rubiaceae	*Scyphiphora hydrophyllacea*	M	Shrub	Chigam
26	Sapotaceae	*Planchonella obovata*	MA	Tree	Menasi
27	Sonneratiaceae	*Sonneratia alba*	M	Tree	Perepat
28	Sonneratiaceae	*S. caseolaris*	M	Tree	Berembang
29	Sonneratiaceae	*S. ovata*	M	Tree	Gedabu
30	Sterculiaceace	*Heritiera littoralis*	MA	Tree	Dungun
31	Leguminosae	*Caesalpinia crista*	MA	Tree	Unak
32	Leguminosae	*Derris trifoliata*	MA	Tree	Tuba laut
33	Leguminosae	*D. uliginosa*	MA	Tree	Setui
34	Malvaceae	*Thespesia populnea*	MA	Tree	Bebaru
35	Pandanaceae	*Pandanus odoratissimus*	MA	Palm	Pandan
36	Tiliaceae	*Brownlowia argentata*	MA	Shrub/tree	Kiei

[a] M: true mangrove; MA: mangrove associate

Silviculture and Carbon Sequestration in Mediterranean Oak Forests

I. Cañellas[1]*, M. Sánchez-González[2], S. M. Bogino[3], P. Adame[4], C. Herrero[5], S. Roig[6], M. Tomé[7], J. A. Paulo[8], and F. Bravo[9]

1 Introduction

The Kyoto Protocol requires every industrialized country to have a transparent and verifiable method for estimating the size and evolution of the carbon stored in forest ecosystems. The intergovernmental panel on climate change (IPCC, 2007) predicts the evolution of the stock over the first commitment period (2008–2012) using the “bottom-up approach”. This approach is based on the use of data from national or regional forest inventories. The biomass of living trees including their dead parts comprises the main carbon pool in forest ecosystems along with the biomass of understorey plants, litter, woody debris and soil organic matter (Pignard et al., 2004).

---

[1] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, *canellas@inia.es

[2] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, msanchez@inia.es

[3] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain

[4] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, adame.patricia@inia.es

[5] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain, chdeaza@pvs.uva.es

[6] Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, sroig@inia.es

[7] Technical University of Lisbon, Instituto Superior de Agronomia, Centro de Estudos Florestais, Lisboa, magatome@isa.utl.pt

[8] Technical University of Lisbon, Instituto Superior de Agronomia, Centro de Estudos Florestais, Lisboa, joanaap@isa.utl.pt

[9] Joint Research Unit INIA-UVa, Department of Forest Resources, University of Valladolid, Palencia, Spain, fbravo@pvs.uva.es

F. Bravo et al. (eds.), *Managing Forest Ecosystems: The Challenge of Climate Change*,

The most recent report on climate change (Ministerio de Medio Ambiente, 2005) and its impact on woodlands in Spain emphasised that the aptitude of woodlands as carbon sinks would increase over the course of the next few decades, but that in the last half of the twenty-first century woodlands could change from being sinks to emitters of carbon dioxide to the atmosphere. This report highlights the urgency of further studies to quantify the total amount of carbon fixed in woodlands and shrublands.

The objective of this chapter is to present some of the studies currently being carried out in Spain and Portugal which are concerned with the possibility of estimating the amount of carbon fixed by two of the main oak species in the Iberian Peninsula; *rebollo* oak (*Quercus pyrenaica* Willd.) and cork oak (*Quercus suber* L.). Three different methodological approaches have been used. The first approach is to use growth models to evaluate the carbon sequestration in both cork and wood over the life of a cork oak plantation. This approach has been applied both for Spain and Portugal. The second approach involves using a yield table as a tool to estimate the carbon sequestration in *Quercus pyrenaica* forests based on Spanish National Forest Inventories. In a third approach, data from a network of plots is used to estimate the carbon sequestration in pure and mixed *Quercus pyrenaica* forests. The application of these different methodologies would allow us to forecast and improve the carbon sequestration in oak forests as well as increasing our understanding of their dynamics.

The two oak species on which this study focuses were chosen because of their ecological and economic importance. At iberian level, cork oak forests occupy more than 1,000,000 ha and in recent times these species have often been used to reforest marginal agricultural areas. Cork oak forests provide a variety of services along with the production of wood and cork. Of these products, cork is a high quality yield. *Rebollo* oak stands occupy almost 600,000 ha and mixed stands are very frequent. Woodlands dominated by *rebollo* oak were very important for firewood production until about 4 decades ago and were an essential element in traditional rural life. In this study we have focused on pure as well as mixed stands of *P. sylvestris* and *Q. pyrenaica*, which occupy more than 160,000 ha in Spain (ICONA, 1998).

2 Biomass Production and Carbon Sequestration in Spanish Cork Oak Forests

The objective of the present work is to estimate the amount of carbon (C) sequestered in Spanish cork oak forests and analyse its evolution during the life cycle of the forest. In this study, we consider the carbon content of wood as well as cork, along with all possible cork extractions made over the rotation period. The following initial hypothesis and restrictions were established:

- The cork oak stand evaluated in the study consisted of a 1 ha plantation aimed mainly at cork production, with trees distributed on a 4 × 3 m^2 design (corresponding to a density of about 825 trees per hectare).
- The site quality of the cork oak forest was medium-good, or Quality II according to Sánchez-González et al. (2005), which corresponds to a dominant height of 12 m at 80 years.
- The debarking period is 10 years, starting from the first cork harvest at 38 years until the end of the rotation, so in total, debarking takes place on thirteen occasions over the rotation.
- At the first cork extraction, the debarking height was 1.2 m, at the second it was 2 m, whilst at the third and consecutive extractions the debarking height was 3 m.
- From an initial plantation density of 825 trees/ha and taking into account natural mortality and thinnings, the density would have been around 500 trees/ha (Montero & Cañellas, 1999) at the first cork extraction, after which the following thinning regime was applied coinciding with different cork harvests:
  - At the second cork harvest, the number of trees per hectare was reduced from 500 to 250.
  - At the fifth, the stand was thinned to 150 trees/ha.
  - At the eighth, the stand was thinned to 70 trees/ha in order to facilitate natural regeneration.
  - Finally, at the eleventh cork harvest, the number of trees per hectare was reduced to 40 trees/ha to promote the establishment of the regeneration.
- It has been assumed that cork produced during the second cork rotation, known as "second cork", can be considered reproduction cork in quantity estimations.
- The rotation is about 150 years, which is considered the upper limit for quality cork production.

We are aware of the limitations of an estimation made under the aforementioned hypothesis but never-the-less believe it to be a valuable exercise which will contribute to our understanding of the role played by these systems in the mitigation of carbon emissions to the atmosphere.

2.1 Material and Methods

Owing to the fact that the C content of wood and cork is different, the dry biomass evolution for cork oak forests in Spain was estimated separately for wood and cork as follows:

2.1.1 Dry Wood Biomass Estimation

Biomass for the whole tree was estimated as the sum of above ground biomass and root biomass using the equations developed by Montero et al. (2005) for cork oak which relate biomass to diameter at breast height under cork:

$$biomass = CF \cdot A \cdot d^b \tag{1}$$

Where $CF = e^{\frac{SSE^2}{2}}$ is a correction factor introduced to help reduce the bias (Sprugel, 1983); and d is the diameter at breast height in cm (under cork for cork oak) (Table 1).

The diameter at breast height under cork was estimated by applying the diameter increment model for cork oak forests (Sánchez-González et al., 2006):

$$\mathrm{idu} = \frac{\mathrm{idu}_{\mathrm{po}}}{1 + \mathrm{e}^{-(0.73+94.97\frac{1}{\mathrm{N}})}} \tag{2}$$

where idu is the annual diameter increment under cork (cm); idu_{po} is the annual potential diameter increment (cm) and N is the number of trees per hectare.

The potential annual diameter increment at breast height under cork was estimated by applying the diameter increment model for dominant trees in cork oak forests (Sánchez-González et al., 2005):

$$du_2 = (83.20 + 5.28SI - 1.53h_1/du_1)^{1-\frac{ln(1-e^{-0.0063t_2})}{ln(1-e^{-0.0063t_1})}} du_1^{\frac{ln(1-e^{-0.0063}}{ln(1-e^{-0.0063}}} \tag{3}$$

where: du_i is the diameter at breast height under cork (cm) at age t_i (years); SI is the site index (m) defined by Sánchez-González et al. (2005) assuming a site index of 12 m corresponding to Quality II and h/d is the height to diameter ratio (cm/cm).

Having calculated dry biomass through Eq. 1, the percentage of Carbon in the whole tree was calculated by multiplying that value by 0.472 (the average carbon content of cork oak wood (Ibañez et al., 2002)), and by the weight ratio of the CO_2 molecule and the Carbon atom 3.67.

2.1.2 Dry Cork Biomass Estimation

Dry cork biomass or cork weight was estimated using the following expression:

$$w = cb \, . \, sh \, . \, cu . \, cork\ density \tag{4}$$

Table 1 Biomass equation parameters for different species and biomass fractions according to Montero et al. (2005)

		Biomass equation parameters		
Species	Biomass fraction	A	B	SSE
Cork oak	*Bt*	0.0346	2.5345	0.41665
	Br	0.0598	2.0777	0.35654
Rebollo oak	*Bt*	0.0745	2.5345	0.2473
	$Bs + Bb_7$	0.0147	2.9597	0.5953
	Bb_2	0.0078	2.3876	0.5629
	Bb_{2-7}	0.1387	1.7730	0.4970
	Br	0.0859	2.1334	0.2429
Pinus sylvestris	*Bt*	0.0818	2.4119	0.2468
	Br	0.0104	2.62841	0.2836

where Bt, Bs, Bb_2, Bb_{2-7}, Bb_7, and Br are total aboveground, stem, branches under 2 cm diameter, branches between 2 and 7 cm diameter, branches above 7 cm diameter and root biomass in kg dry matter (D.M.) respectively. SEE, standard error of the estimate

where *w* is cork weight (kg D.M.), *sh* is stripped height (m); *cu* is circumference at breast height under cork (m) calculated from diameter at breast height under cork; *cork density* was assumed to be 315 kg/m^3 for dry cork (Pereira, 2007) and *cb* is predicted cork thickness estimated using the following cork growth model (Sánchez-González et al., 2007):

$$cb_2 = cb_1 \left(\frac{1-e^{-0.04t}}{1-e^{-0.04t_0}} \right)^{\frac{0.57+1.86}{X_0}} \tag{5}$$

where: $X_0 = \frac{1}{2}\left(\begin{array}{c} \left(ln\left(cb_1\right) - 0.57ln\left(1-e^{-0.04t_0}\right)\right) \pm \\ \sqrt{\left(ln\left(cb_1\right) - 0.57ln\left(1-e^{-0.04t_0}\right)\right)^2 - 4\cdot 1.86ln\left(1-e^{-0.04t_0}\right)} \end{array} \right)$ cb_i is cork thickness (cm) at age t_i (years).

Only data from complete growth years were considered in the cork growth model (3,745 observations from 432 trees), the first half year and last half year of a debarking period were not considered. The variable modelled is then the accumulated cork thickness after *t* complete years of growth. When calculating cork biomass and values for CO_2 sequestration in cork, we have added the mean value for the first half year and last half year of a debarking period to the cork thickness corresponding to the last complete year before cork harvest.

Having obtained a value for cork weight, the percentage of Carbon in cork was calculated by multiplying that value by 0.57 (according to Gil et al., 2005 the average carbon content of cork is 57%), and by the weight ratio of the CO_2 molecule and the Carbon atom, 3.67.

2.2 Results and Conclusion

The wood biomass production in cork oak forests shows an increasing trend (Fig. 1 and Table 2). The wood biomass extracted at each thinning totals 297.46 Mg/ha. It should be mentioned that at the first cork extraction, the amount of cork biomass extracted is zero because this is "virgin cork" which has different characteristics to the cork extracted in successive harvests which is termed "reproduction cork". Regarding cork biomass production (Fig. 2 and Table 2), the maximum value is reached at 88 years, coinciding with the sixth cork harvest.

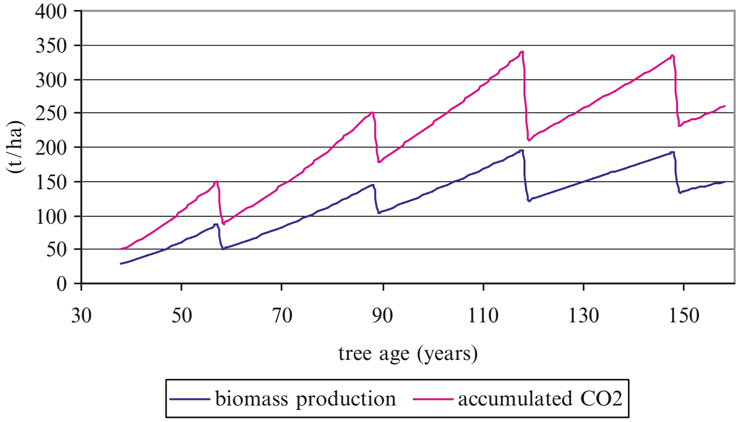


Fig. 1 Biomass production and accumulated CO_2 values for wood in cork oak forests

Table 2 Wood and cork biomass production in cork oak forests

Age (years)	du	cu	Trees/ha	WB	CB
38	14.61	63	500	28.82	0.00
48	18.93	76	500	54.52	4.21
58	24.39	94	250	51.06	7.59
68	28.77	107	250	76.96	7.15
78	33.01	121	250	108.45	8.08
88	37.11	134	150	145.24	8.97
98	43.45	153	150	129.40	6.61
108	47.45	166	150	161.33	7.63
118	51.26	178	70	195.94	4.09
128	61.33	210	70	143.65	4.70
138	65.20	222	70	167.62	4.94
148	68.88	233	40	192.53	5.17
158	77.86	262	40	149.97	3.69

du: diameter at breast height under cork (cm); cu: circumference at breast height over cork (cm); WB: wood biomass (t D.M./ha); CB: cork biomass (t D.M./ha)

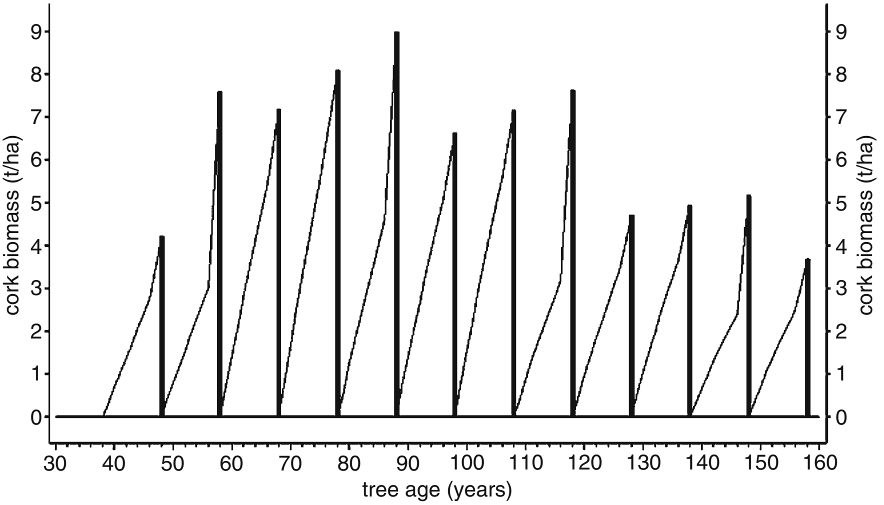


Fig. 2 Cork biomass production in cork oak forests. The line shows cork biomass evolution, and the bars represent the cork biomass extracted at each harvest

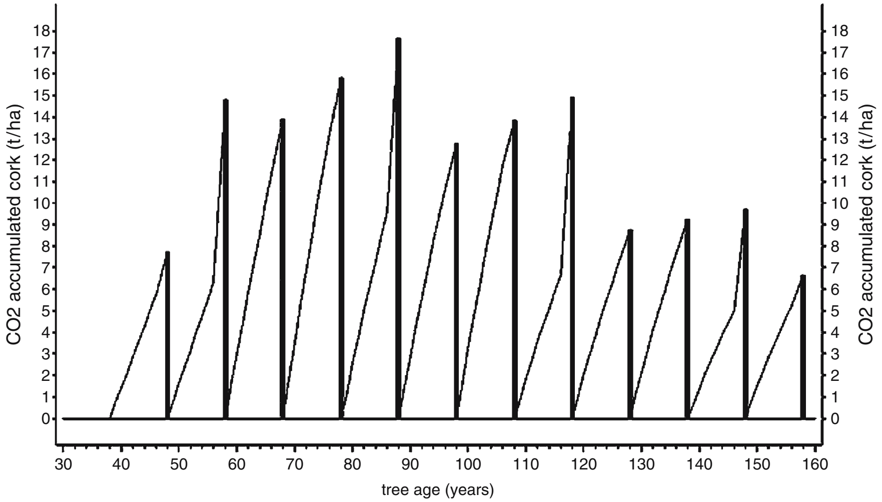


Fig. 3 Accumulated CO_2 values at each cork harvest (line) and amount of CO_2 extracted in each (bars)

As the carbon content of cork oak wood and cork is different, when evaluating the amount of CO_2 stored in cork oak forests it is important to differentiate between the CO_2 accumulated in wood (Fig. 1) and in cork (Fig. 3). In the case of cork, the accumulated CO_2 is extracted at each cork harvest although it remains sequestrated in cork products. The total amount of accumulated CO_2, considering all the cork extractions, reaches 145.60 Mg D.M./ha.

The amount of carbon stored in a cork oak forest with the characteristics described in the introduction section would be 452.56 Mg D.M./ha for a rotation of 158 years. This figure gives an idea of the importance of these stands in the mitigation of carbon emissions to the atmosphere. Nevertheless, it is necessary to improve these calculations through continued research into the growth and yield of these forests, in particular, those associated with virgin cork.

3 Biomass Production and Carbon Sequestration in Portuguese Cork Oak Forests

Cork oak is a very important species in Portugal. According to the last National Forest Inventory (Tomé et al., 2007) the species covers an area close to 680 10^3 ha (pure stands and stands with cork oak as dominant species). A great majority of the present stands are adult stands characterised by a low density, average stand density in the country is less than 60 trees per hectare and 85% of the stands have less than 120 trees per hectare. New plantations began to be established by the end of the 1980s and nowadays they represent an important area in the country. According to official statistics (Ferreira et al., 2006) rates of planting have been of the order of 4,800 ha/year and 9,300 ha/year for the periods 1990–1994 and 1995–2003, respectively. The objective of the present work is to estimate the carbon sequestered by a stand of average site index and, taking the rates of plantation into account, to estimate the amount of carbon that is expected to be sequestered by the plantations established after 1990 (article 3.3 of the Kyoto protocol) during the first commitment period of 2008–2012.

3.1 Material and Methods

The estimates of the evolution of carbon stocks in cork oak stands were based on the version 3 of the SUBER model (Tomé, 2004) to which the system of equations for tree biomass estimation developed by Paulo and Tomé (2006) was added. Crown width was estimated with the equations developed for the Portuguese NFI 2005/2006 (Tomé et al., 2007) and root biomass with the equation developed for Spain by Montero et al. (2005) presented in Table 1.

3.1.1 Developing the Yield Table

Figure 4 represents the site index curves developed by Sánchez-González et al. (2005) for Spain and that have been tested with success for Portugal (Tomé, 2004) jointly with the information from the permanent plots included in the data base SUBER-DATA for cork oak growth in Portugal (Coelho & Godinho, 2002) for

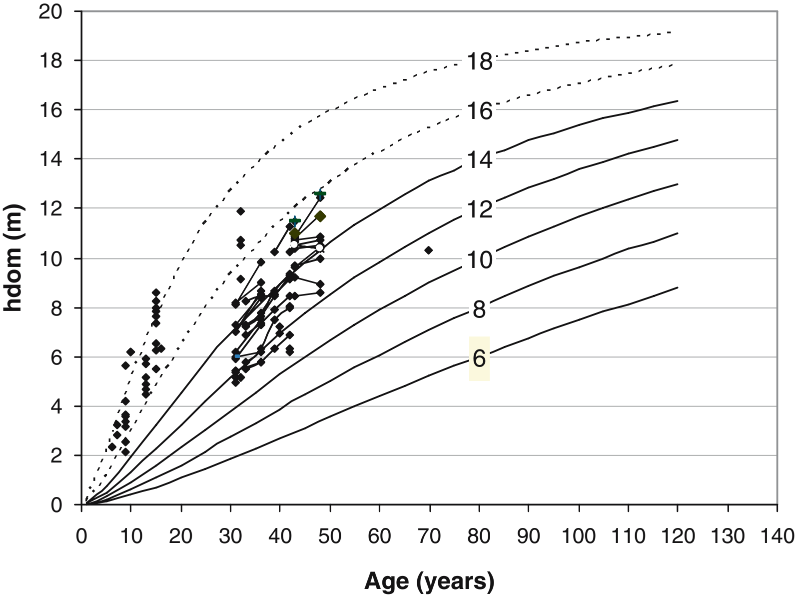


Fig. 4 Comparison between the Spanish site index curves for cork oak (Sánchez-González et al., 2005) and the permanent plots established in Portuguese stands in which the age is known. Observations from the same plots are connected with straight lines

which the age is known. The figure clearly shows that the new plantations (younger than 20 years) were established in better sites that the older ones. For this study new plantations were assumed to have an average the site index of 16 m (base age 80). The SUBER model was run in order to build a yield table for a stand with site index equal to 16 m. The following assumptions were made for the development of the yield table:

- Initialization was made with a simulated stand of 15 years planted. It was assumed that the stand was planted with 500 trees per hectare and that initial mortality till this age was of 15%; diameter distribution was simulated in order to be similar to the one observed in the permanent plots established in young plantations with a site index close to the average site index for new plantations (16 m).
- The debarking period was assumed to be 9 years, starting when the quadratic mean diameter of the stand was greater than 25 cm (according to Portuguese legislation cork can not be extracted before the perimeter at breast height attains 70 cm).
- The debarking coefficients (ratio between debarking height and tree perimeter at breast height) were: 2 for the first cork extraction, 2.5 for the second cork extraction and 3 for the third and consecutive extractions.
- Stand density was defined by thinning every time the percent crown cover at the year of debarking was higher than 50%. This percent crown cover is close to the

maximum recommended by Natividade (1950). Stands were thinned using the thinning algorithm presented in Tomé (2004).

- It has been assumed that cork produced during the second cork rotation, known as "second cork", can be estimated with the equations available for reproduction cork.
- The rotation considered was 150 years, which is considered the upper limit for quality cork production.

3.1.2 Simulation of Future Total Carbon Stocks

The simulation of the future carbon stocks was based on the estimation, for each year, of the areas of new plantations by age class (age classes with 1 year were used). According to official statistics (Pereira, 2006) rates of planting have been of the order of 4,800 ha/year and 9,300 ha/year for the periods 1990–1994 and 1995–2003, respectively. This last rate was assumed for the years after 2003. This information was used to estimate, for each year, the area of new plantations in age class 1 (between 0 and 1). The area in age class *j* in year *t + 1* was estimated as 98% of the area in age class *(j − 1)* in year *t*, which means an annual loss rate of 2% of the area mainly due to forest fire and land use change. An additional annual loss of 5% was assumed for stands with an age less than 6 years to consider the non-success of some plantations. Total carbon stock at year t was estimated as:

$$Cstock_t = \sum_{j=1}^{150} \left[0.472\left(Bw_j + Br_j\right) + 0.57\,Bb_j + 0.5\,Bc_j \right] A_{j,t}, \tag{6}$$

where *Bwj, Brj, Bbj and Bcj* are, respectively, stem wood, root, bark (cork) and crown biomass per ha at age j as given by the yield table and *Aj,t* is the area of new plantations in age class *j* in year *t*.

3.1.3 Estimation of Carbon Sequestration During the Period 2008–2012

Carbon sequestered during the first commitment period of the Kyoto protocol (2008–2012) was estimated by the stock change approach:

$$\Delta C = Cstock_{2013} - Cstock_{2008}, \tag{7}$$

where $Cstock_t$ is the carbon stock at the start of year *t*.

3.2 Results and Discussion

The evolution of a cork oak plantation with the characteristics described in Section 3.1.1 is presented in Fig. 5. It is important to point out that the traditional cork oak stands are not managed with such a high percent of crown cover as they are managed

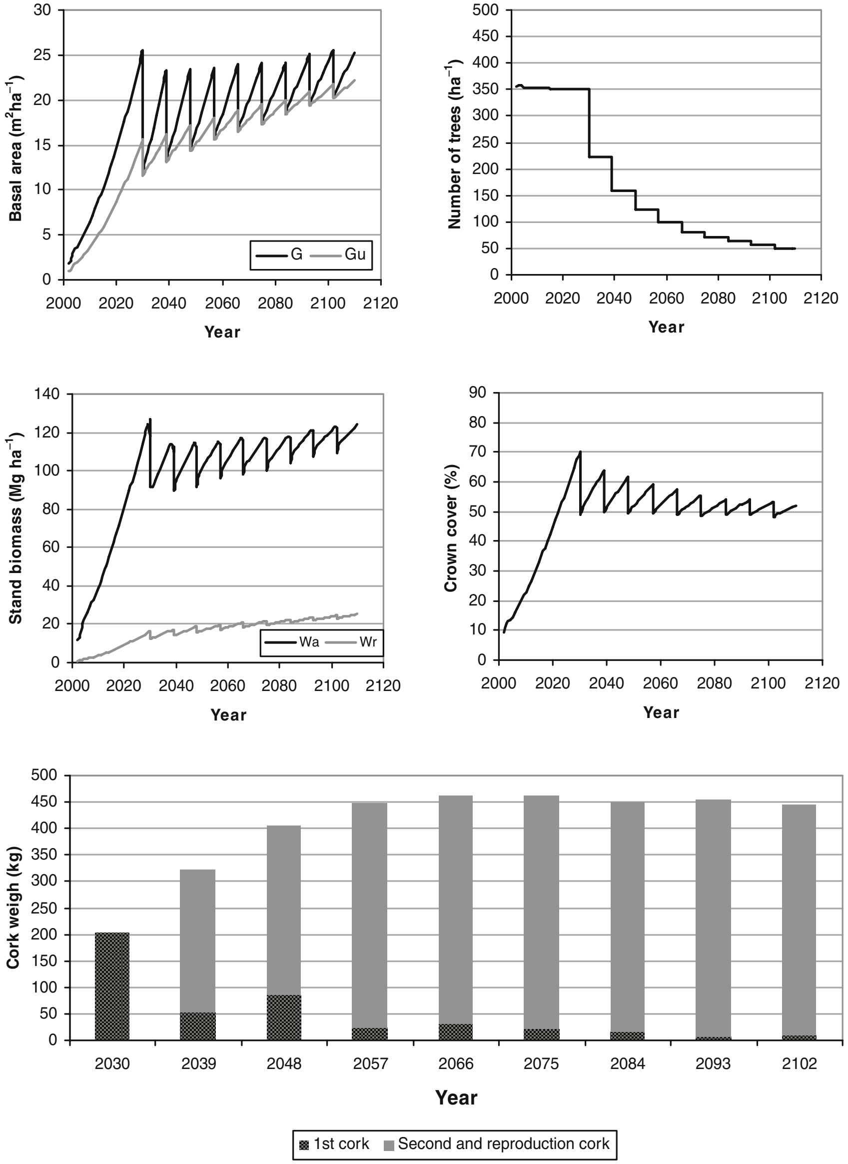


Fig. 5 Evolution of a new cork oak plantation with the characteristics presented in 3.1.1. G and Gu are, respectively, over and under bark basal area and Ba and Br are total aboveground and root biomass

as an agro-silvopastoral system that require low crown cover compatible with grazing underneath. However most of the new plantations are being managed with cork production as the main objective. This option can only be achieved with

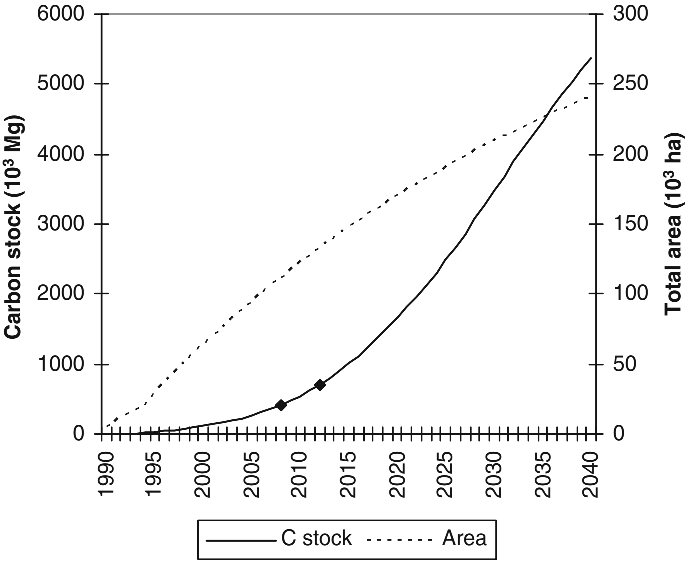


Fig. 6 Estimation of the evolution of the area of new cork oak plantations and the corresponding fixation of carbon for the period 1990–2040. The black lozenges indicate the years 2008 and 2013

a complete use of the space by the trees, therefore a high percent of crown cover was used in the development of the yield table.

Figure 6 shows the estimations of total area and the corresponding carbon stock for the first 50 years (till 2040). According to the assumptions used for the development of the yield table (see Section 3.1.1), new plantations of cork oak may sequester as much as 5.9 Mt during a period of 50 years. However the amount of C sequestered during the first commitment period of the Kyoto protocol is just 0.42 Mt of C sequestered that is equivalent to 1.4 Mt of CO_2 equivalent. This value represents a large share of the value estimated by Portugal for this period for the new plantations of all species.

This result is very important for Portugal as recently (October 2007) the European Commission has reduced the estimations of Portugal for reductions induced by afforestation and reforestation due to the fact that the Portuguese estimates were not well justified. The results presented here support the initial predictions of Portugal indicating that the value presented is not an over estimation but, on the contrary, may be an under estimation. Even if the estimations are based in a lower site index (14 m) the estimation of the reductions induced by new cork oak plantations during the 2008–2012 period will be 0.7 Mt of CO_2 equivalent.

4 Carbon Quantification in Pure *Quercus pyrenaica* Willd. Woodlands in Spain

In the Iberian Peninsula, traditional uses of *rebollo* oak stands have been progressively abandoned over the last 4 decades. Today, these woodlands suffer from a number of silvicultural, ecological and socioeconomic problems (Cañellas et al., 2004) which often means that they do not meet the minimum requirements for the application of silvicultural treatment programmes. Nevertheless, the role of *Rebollo* oak woodlands as carbon stores should be taken into consideration when developing management plans. Given the widespread distribution of the species and current lack of demand for direct products which would justify the application of intensive silviculture (bearing in mind the high cost of such treatments), it is necessary to prioritize the management of these stands in accordance with the available resources and final objectives. *Rebollo* woodlands with medium to high densities and regular diameter distributions present critical problems of stagnation and fire risk and therefore should be considered a priority when applying silvicultural treatments.

Growth and yield models for variable silviculture indicate the silvicultural interventions to be carried out during the rotation, the approximate age of the stand in which these should be performed, and the results which may be expected according to the intensity of the interventions (Cañellas et al., 2000). Models of this type do not follow a strict procedure or a fixed set of rules, but sometimes provide very detailed guidelines for silvicultural interventions (Montero et al., 2001).

The aim of this study is to quantify the current stocks and flows of carbon in the main Spanish *rebollo* stands (coppices) in which thinning treatments might be of interest. For the purposes of the study, four different site qualities and two silvicultural schemes were considered. The latter would be more intensive in the better quality sites and moderate in the lower quality sites.

4.1 Data and Methods

The data were obtained from 200 plots belonging to the Spanish National Forest Inventory (SNFI) (DGCN, 1996) in the northwest area of Spain (Castilla y León region). According to Elena Roselló (1997), the region is divided into four ecoregions defined according to biogeoclimatic conditions. The plots were selected in proportion to the extension of the different ecological strata, providing representative stands with a variety of stand structure and site conditions.

Yield models for *Q. pyrenaica* in northwest Spain were built to simulate the different silviculture alternatives and predict changes in the basic stand variables: number of stems per hectare (N), quadratic mean diameter (Dg), basal area (G) and volume (V) for a wide range of density management regimes. Natural mortality was considered to be null as it was controlled by thinning. The structure of these models

Table 3 Thinning regimes applied in rebollo coppices

Intensive silvicultural treatments	Moderate silvicultural treatments
≈20 years: if initial density higher 2,000–2,500 trees per hectare pre-commercial thinning has to be applied	≈15–20 years: if initial density higher 3,500–4,000 trees per hectare pre -commercial thinning has to be applied
30 years: 1st low thinning	30 years: 1st low thinning
40 years: 2nd low thinning	50 years: 2nd low thinning
60 years: 3rd low thinning	60 years: 3rd low thinning
80 years: 4th low thinning	80 years: 4th low thinning
Rotation: 100–120 years	Rotation: 100–120 years

is based on equations involving site index (*SI*), stems per hectare (N), volume (*V*), and mean height (*Hm*):

- 1st equation: site index defined by Adame et al.(2006) (8)
- 2nd equation: $N = \exp(8.49459 - 0.114019 \cdot Ho)$ (9)

  $R^2 = 0.37$

- 3rd equation: $Dg = 2.17047 + \frac{2174.69}{N^{0.7415}} + 0.215575 \cdot Ho$ (10)

  $R^2 = 0.95$

- 4th equation: $\ln(V) = -0.709618 + 0.936607 \cdot \ln(G \cdot Ho)$ (11)

  $R^2 = 0.98$

- 5th equation: $Hm = 1.20895 + 0.643414 \cdot Ho$ (12)

  $R^2 = 0.64$

where *N*: is the number of stems per hectare, *Ho*: dominant height (m), *Dg*: quadratic mean diameter (cm), *V*: stem volume per m^3/ha, *G*: basal area (m^2/ha), *Hm*: mean height.

Intensive silvicultural treatments (Site index 16 and 13 according Adame et al., 2006) and moderate silvicultural treatments (Site index 10 and 7) are presented in Table 3. The yield table according to site quality and silvicultural treatments is shown in Table 4.

The dry biomass of the whole tree was estimated as the sum of the different fractions (stem, branches with a diameter over 7 cm, branches with a diameter of between 2 and 7 cm, branches with a diameter of less than 2 cm including needles and roots) using the equation developed by Montero et al. (2005) for *Quercus pyrenaica* Willd (Eq. 1 and Table 1).

Table 4 Yield tables according to site quality and silvicultural treatments.

SITE INDEX 16: INTENSIVE SILVICULTURAL TREATMENTS

		Main crop before thinning				Crop removed			Main crop after thinning			
Age (years)	*Ho* (m)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)
30	10.7	2,000	12.2	23.5	87.5	650	10.4	19.4	1,350	13.0	18.0	68.1
40	13,0	1,350	15.4	25.0	110.7	500	13.0	28.0	850	16.6	18.3	82.7
60	16.9	850	20.2	27.4	146.4	300	17.2	35.3	550	21.7	20.4	111.2
80	17.8	550	26.2	29.6	174.0	150	22.3	32.3	400	27.5	23.8	141.7
100	18.8	400	31.8	31.8	196.2							

SITE INDEX 13: INTENSIVE SILVICULTURAL TREATMENTS

		Main crop before thinning				Crop removed			Main crop after thinning			
Age (years)	*Ho* (m)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)
30	7.9	2,500	10.4	21.4	60.0	700	8.7	10.9	1,800	11.1	17.3	49.1
40	10.0	1,800	12.7	22.9	79.8	600	10.5	17.3	1,200	13.7	17.6	62.5
60	13.0	1,200	16.3	25.0	110.9	400	13.5	24.0	800	17.5	19.3	86.9
80	14.8	800	20.7	26.8	133.7	300	17.1	32.6	500	22.5	19.9	101.1
100	15.9	500	27.3	29.2	155.2							

SITE INDEX 10: MODERATE SILVICULTURAL TREATMENTS

		Main crop before thinning				Crop removed			Main crop after thinning			
Age (years)	*Ho* (m)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)
30	5.3	3,400	8.6	19.5	38.2	1,000	6.4	6.8	2,400	9.3	16.3	32.3
50	8.7	2,400	10.8	22.1	68.2	750	8.1	11.3	1,650	11.9	18.2	56.9
70	10.9	1,650	13.5	23.6	89.5	600	10.1	17.3	1,050	15.1	18.7	72.3
90	12.4	1,050	17.3	24.8	105.3	350	13.0	18.6	700	19.2	20.2	86.7
120	13.6	700	22.0	26.6	122.7							

SITE INDEX 7: MODERATE SILVICULTURAL TREATMENTS

		Main crop before thinning				Crop removed			Main crop after thinning			
Age (years)	*Ho* (m)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*V* (m^3/ha)	*N* (trees/ha)	*Dg* (cm)	*G* (m^2/ha)	*V* (m^3/ha)
30	3.1	4,000	7.49	17.6	21.1	1,200	5.62	3.36	2,800	8.2	14.6	17.8
50	5.9	2,800	9.49	19.8	42.5	900	7.12	7.25	1,900	10.4	16.2	35.3
70	7.9	1,900	11.9	21.2	59.6	650	8.95	10.8	1,250	13.2	17.1	48.8
90	9.2	1,250	15.1	22.5	72.8	400	11.4	12.3	850	16.6	18.5	60.4
120	10.4	850	19	24.2	87.1							

Once dry biomass has been estimated, the percentage of carbon in each fraction and in the tree as a whole was calculated by multiplying each value by 0.475 according to Ibáñez et al. (2002).The carbon stored in the soil organic matter is not considered in this study.

4.2 Results and Discussion

The results obtained demonstrate the relevance of both site quality and forest management in the preservation of forests as carbon sinks. The carbon stocks in each biomass fraction of the *Q. pyrenaica* in northwest Spain are given in Tables 5 and 6. Figure 7 shows the total carbon fixed (in tons) taking into account site index and age.

The results for carbon accumulation in *rebollo* oak (Tables 5 and 6) show that the better the quality, the higher the total amount of carbon fixed since the biomass volume increases with quality. For example, the current increment in total carbon fixed at an average age of 60–80 years for site index 16 is 3.81 Mg/ha-year, 2.84 Mg/ha-year for site index 13, 2.02 Mg/ha-year for site index 10 and 0.82 Mg/ha-year for site index 7.

Total fixed carbon on site index 16 represents a 18.5% more than it is fixed on site index 13, a 53.6% more than it is fixed on site index 10 and 80.8% more than it is fixed on site index 7. Relevance of forest management can be seen on final differences in total carbon sequestration (Fig. 7). Meanwhile, initial differences at 30 years between intensive and moderate silvicultural treatments ranged between 61.9% (between site index 16 and site index 7) and 16.6% (between site index 13 and 10), final differences at 100 years in total carbon fixed (the combination of carbon removed and carbon total) ranged between 80.8% and 29.7% respectively.

5 Carbon Quantification in Pure and Mixed *Quercus pyrenaica* Stands in Central Spain

The objective of this section is to estimate the amount of carbon dioxide fixed by pure and mixed woodlands of *Quercus pyrenaica* located in the Spanish Central Mountain Range using inventory data from temporary plots. A more complete understanding of the situation might help forest managers to adapt the management of these stands according to their specific composition and thus increase their capacity as carbon sinks.

5.1 Materials and Methods

Permanent plot data from Valsaín forest (Central Spain) were used to estimate carbon sequestration by oak stands (Table 7). Three plots ($50 \times 50\,m^2$) in each of the forest types (mixed and pure oak woodland) were established. The total, above and below-

Table 5 Carbon concentrations (t/ha) in different components of rebollo oak for high quality sites

SITE INDEX 16 INTENSIVE SILVICULTURAL TREATMENTS

	Carbon before thinning						Carbon removed						Carbon after thinning					
Age (years)	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT
30	41.36	27.34	12.58	3.42	17.47	58.83	8.97	5.54	3.08	0.76	4.04	13.01	32.79	22.27	9.50	2.68	13.50	46.30
40	50.38	36.77	12.83	4.02	19.38	69.76	12.15	8.25	3.52	0.99	5.00	17.15	38.37	28.91	9.23	3.03	14.32	52.69
60	63.10	51.68	13.07	4.84	21.77	84.87	14.82	11.33	3.47	1.16	5.45	20.27	48.95	41.34	9.60	3.72	16.41	65.37
80	78.93	72.21	13.41	5.83	24.54	103.46	14.31	12.22	2.75	1.08	4.74	19.05	64.90	60.61	10.63	4.76	19.79	84.68
100	93.79	93.17	13.75	6.73	26.98	120.76												

SITE INDEX 13: INTENSIVE SILVICULTURAL TREATMENTS

	Carbon before thinning						Carbon removed						Carbon after thinning					
Age (years)	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT
30	34.50	21.31	11.85	2.92	15.53	50.03	6.14	3.52	2.42	0.53	2.97	9.12	29.30	18.60	9.58	2.45	12.85	42.15
40	41.21	27.71	12.16	3.38	17.13	58.34	8.48	5.26	2.89	0.72	3.81	12.29	33.29	23.12	9.27	2.70	13.42	46.72
60	51.72	38.67	12.62	4.09	19.45	71.17	10.69	7.38	3.01	0.87	4.34	15.03	41.28	31.81	9.54	3.23	15.09	56.37
80	63.18	52.30	12.85	4.83	21.59	84.77	14.60	11.14	3.43	1.15	5.39	19.98	48.78	41.83	9.31	3.68	16.12	64.90
100	79.63	74.14	13.12	5.84	24.35	103.98												

where Bt, Bs, Bb_2, Bb_{2-7}, Bb_7, and Br are total aboveground, stem, branches under 2 cm diameter, branches between 2 and 7 cm diameter, branches above 7 cm diameter and root biomass in kg D.M. respectively. BT is the total tree carbon fixed in Mg/ha

Table 6 Carbon concentrations (t/ha) in different components of rebollo oak for low quality sites

SITE INDEX 10: MODERATE SILVICULTURAL TREATMENTS

Age (years)	Carbon before thinning						Carbon removed						Carbon after thinning					
	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT
30	28.98	16.51	11.50	2.52	14.08	43.07	4.03	2.03	2.00	0.37	2.21	6.24	24.95	14.69	9.33	2.14	11.75	36.69
40	36.44	22.87	12.16	3.06	16.16	52.60	5.49	3.05	2.28	0.48	2.73	8.23	32.03	20.95	9.93	2.66	13.67	45.70
60	44.10	30.44	12.42	3.59	17.89	61.99	7.69	4.69	2.70	0.65	3.50	11.19	37.28	26.98	9.64	2.98	14.46	51.74
80	52.62	40.36	12.27	4.13	19.32	71.95	8.50	5.77	2.46	0.70	3.50	12.00	45.69	36.63	9.84	3.53	16.09	61.78
100	64.51	54.80	12.52	4.89	21.51	86.02												

SITE INDEX 7: MODERATE SILVICULTURAL TREATMENTS

Age (years)	Carbon before thinning						Carbon removed						Carbon after thinning					
	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT	Bt	$Bs + Bb_7$	Bb_{2-7}	Bb_2	Br	BT
30	24.00	12.89	10.59	2.13	12.33	36.33	3.47	1.65	1.91	0.32	2.00	5.47	20.88	11.64	8.63	1.83	10.37	31.25
40	30.61	18.18	11.28	2.62	14.30	44.91	4.75	2.49	2.18	0.42	2.49	7.23	26.37	16.30	9.04	2.23	11.86	38.23
60	37.11	24.30	11.48	3.07	15.82	52.92	6.12	3.55	2.36	0.53	2.93	9.05	31.66	21.65	9.06	2.58	12.95	44.60
80	44.72	32.41	11.54	3.58	17.32	62.04	6.90	4.43	2.22	0.58	3.00	9.90	38.55	29.07	9.26	3.04	14.38	52.93
100	54.28	43.36	11.77	4.20	19.18	73.46												

where Bt, Bs, Bb_2, Bb_{2-7}, Bb_7, and Br are total aboveground, stem, branches under 2 cm diameter, branches between 2 and 7 cm diameter, branches above 7 cm diameter and root biomass in kg D.M. respectively. BT is the total tree carbon fixed in Mg/ha

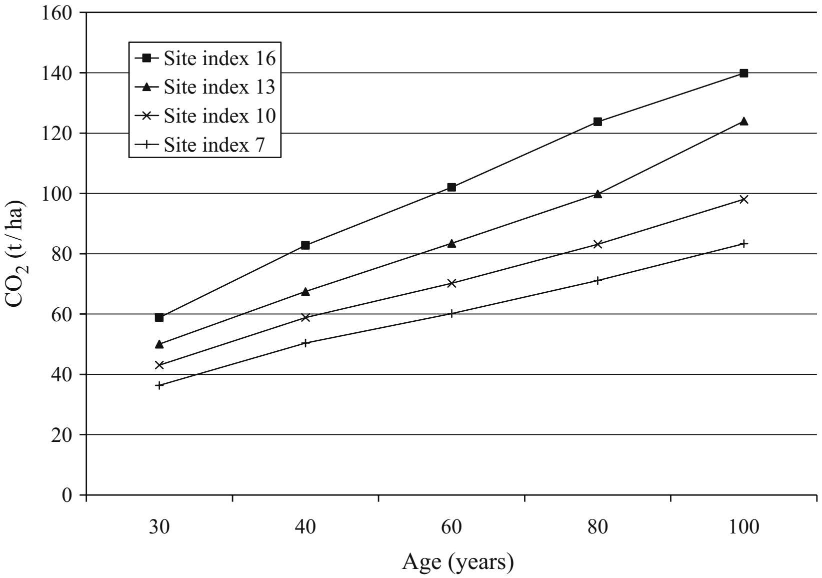


Fig. 7 CO_2 sequestration in relation to age and site index in rebollo oak coppice stands in north-west Spain

Table 7 Main characteristics of sampled plots

	Mixed stands (n = 3)		Oak stands (n = 3)	
	Mean	SD	Mean	SD
N	846.67	367.75	622.67	200.57
BA	33.22	12.02	13.86	1.93
Dg	23.06	4.92	17.1	1.52
Dbh	19.74	3.67	15.87	0.68
CO_2 total	311.34	143.46	122.19	8.51
CO_2 above	260.74	110.31	116.47	8.46
CO_2 below	37.52	19.74	5.72	0.304

N: number of trees (tress/ha); BA: Basal area (m^2/ha); Dg: Quadratic mean diameter (cm); dbh: Mean diameter at breast height (cm), CO_2 total: total amount of carbon dioxide (t/ha); CO_2 above: total amount of carbon dioxide aboveground (Mg/ha); CO_2 below: total amount of carbon dioxide below-ground (Mg/ha)

ground dry biomass by fraction (stem, roots, branches and needles) were estimated using the Montero et al. (2005) models (Table 1). Carbon content was estimated as 0.475 of biomass, according to Ibáñez et al. (2002) and IPCC (2007) guidelines.

A two-way Analysis of Variance (ANOVA) was conducted to test if there were significant differences between stands. The factors included in the model were forest structure (coefficient of variation of diameter at breast height distribution) and dominant species (oak and mixed woodlands) while the response considered was the total, above and below-ground amount of fixed carbon. The coefficient of variation was previously classified into three classes: coefficients less than 35% were considered as homogeneous stands, coefficients more than 45% were considered as heterogeneous stands and coefficients between 35% and 45% were considered as intermediate stands.

5.2 Results and Conclusions

Significant differences were found between the total carbon dioxide stock and the coefficient of variation of the mean diameter and specific composition. Factors included in the total carbon dioxide model did not interact. The results show that mixed stands of *Quercus pyrenaica* fixed 311.34 Mg/ha of carbon dioxide and pure woodlands fixed 122.19 Mg/ha (Table 7). As in the total carbon dioxide analysis, significant differences were found between the aboveground fixed carbon dioxide and the coefficient of variation of the mean diameter and specific composition. Factors included in the model did not show interaction. As in the total fixed carbon dioxide results, the mixed stand fixed a higher amount of aboveground carbon dioxide (260.74 Mg/ha) than the pure woodlands of *Quercus pyrenaica* (116.47 Mg/ha). The belowground carbon dioxide sequestration analysis reveals that forest structure is not of importance (p = 0.4093) whereas the specific composition has a significant influence (p = 0.0398). The results reveal no significant differences between pure *Quercus pyrenaica* and mixed woodlands (Table 7).

6 Final Remarks

Management objectives and techniques in woodlands dominated by species of Quercus have changed dramatically over the last few decades, especially in the case of coppices. The fact that these stands are of little economic importance today is in part compensated by their value, for example, as carbon sinks. The identification of this important role of Quercus woodlands allows us to contemplate the development of specific silviculture and to promote investment in the stands. It is also important, in the context of global change, to analyze the dynamics involved in the transformation of *Quercus* woodlands from carbon sinks to emitters.

In this chapter we have applied three different methodologies to calculate the biomass and C storage in woodlands composed of two different *Quercus* species. Although some important results have been obtained and analyzed, there are several issues which need to be addressed in future research, such as the development of new biomass and C equations for the different elements of the ecosystem, the role of the root systems in the C cycle, the influence of management techniques on biomass or C storage, the influence of management on the specific composition, the response of stands (coppices) with intensive silviculture regimes to changes in management practices and the effect of new stand uses on C sequestration, etc.

It is difficult to compare the results with those of previous studies because of the differences in forest types, site conditions, management systems, monitoring methodology or time scales. Furthermore, this study focuses exclusively on that part of the carbon cycle in forest ecosystems related to wood production. In previous studies, carbon sequestration related to afforestation, agro forestry and forest management projects (Masera et al., 2003) was modelled to estimate the rate of variation of carbon dioxide in different ecosystems. The authors concluded that forest stands of Norway Spruce, *Picea abies* (L.) Karst, in Northern Europe fixed 120 Mg/ha of carbon dioxide in 100 year old plantations. Even aged beech forest in Atlantic Europe, during the same period, fix 150 Mg/ha. Although the species analysed were different from those of this study, the results are similar to those for pure *Q. pyrenaica* woodlands. Other studies (Lal, 2004) have indicated that woodlands growing in tropical areas can restore 170 Mg/ha over a much shorter period of 50 years.

It is currently believed that not only the biomass but also the soil capacity is relevant to carbon dioxide capture and fixation. Previous studies developed in temperate and boreal woodlands (Patenaude et al., 2004) suggest that forest soils can capture similar amounts of carbon dioxide to that captured by the trees growing in our study site. Therefore, the inclusion of carbon dioxide sequestration by soils would appear to be an important element for future research. However, one of the greatest weaknesses of this kind of study is the lack of information about the belowground biomass.

References

Adame Hernando, P., Cañellas, I., Roig, S., del Rio, M., 2006. Modelling dominant height growth and site index curves for rebollo oak (*Quercus pyrenaica* Willd.). Ann. For. Sci. 63, 929–940.

Cañellas, I., Martínez, F., Montero, G., 2000. Silviculture and dynamics of *Pinus sylvestris* L. stands in Spain. Investigación Agraria. Sistemas y recursos forestales Fuera de serie, 233–254.

Cañellas, I., del Río, M., Roig, S., Montero, G., 2004. Growth response to thinning in *Quercus pyrenaica* Willd. coppice stands in Spanish central mountain. Ann. For. Sci. 61, 243–250.

Coelho, M.B., Godinho, J.M., 2002. SUBER DATA 2 - Base de dados de sobreiro - Manual para o utilizador. Publicações GIMREF - RT8/2002. Departamento de Engenharia Florestal, Instituto Superior de Agronomia, Lisboa.

DGCN, 1996. II Inventario Forestal Nacional 1986–1996. Direccion General de Conservación de la Naturaleza, Ministerio de Medio Ambiente, Madrid.

Elena Roselló, R., 1997. Clasificación biogeoclimática de España Peninsular y Balear. MAPA, Madrid.

Ferreira, V.G., Pereira, T.C., Seabra, T., Torres, P., Maciel, H., 2006. Portuguese national inventory report on greenhouse gases, 1990–2004 – submitted under the United Nations Framework on Climate Change. Ministry of the Environment and Landuse Planning, Institute for the Environment, Amadora.

Gil, L., Pereira, C., Silva, P., 2005. Cork and CO2 fixation. Proceedings of SUBERWOOD: New challenges for integration of cork oak forests and products. Universidad de Huelva, Huelva, 20–22 de octubre de 2005.

Ibáñez, J.J., Vayreda, J., Gracia, C., 2002. Metodología complementaria al Inventario Forestal Nacional en Catalunya. In: Bravo, F., del Río, M., del Peso, C. (Eds.), El Inventario Forestal Nacional. Elemento clave para la gestión forestal sostenible. Fundación General de la Universidad de Valladolid, pp. 67–77.

ICONA, 1998. Segundo Inventario Forestal Nacional, España. Ministerio de Medio Ambiente, Dirección General de Conservación de la Naturaleza, 337 p.

IPCC, 2007. Intergovernmental panel on climate change 2007. www. ipcc2007.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1–22.

Masera, O., Garza-Caligaris, F., Kanninen, M., Karjalainen, T., Liski, J., Nabuurs, G., Pussinen, A., De Jong, B., Mohren, G., 2003. Modeling carbon sequestration in afforestation, agroforestry and forest management projects: The CO2FIX V.2 approach. Ecol. Model. 164, 177–199.

Ministerio de Medio Ambiente, Universidad de Castilla-La Mancha, 2005. Evaluación preliminar de los impactos en España por el efecto del cambio climático, p. 846.

Montero, G., Cañellas, I., 1999. Manual de forestación del alcornoque (*Quercus suber* L.). MAPA-INIA, 103 p.

Montero, G., Cañellas, I., Ruíz-Peinado, R., 2001. Growth and yield models for *Pinus halepensis* Mill. Investigación Agraria. Sistemas y Recursos Forestales 10, 179–201.

Montero, G., Ruiz-Peinado, R., Muñoz, M., 2005. Producción de biomasa y fijación de C02 por los bosques españoles. Monografías INIA, Seria Forestal no. 13, 270 p.

Natividade, J., Vieira, 1950. Subericultura. Ministério da Economia, Direcção Geral dos Serviçoes Florestais e Aquícolas, Lisboa.

Paulo, J.A., Tomé, M., 2006. Equações para Estimação do Volume e Biomassa de Duas Espécies de Carvalhos: Quercus suber e Quercus ilex. Relatório científico do GIMREF, no. 1/2006. Centro de Estudos Florestais, Instituto Superior de Agronomia, Lisboa.

Patenaude, G., Hill, R., Milne, R., Gaveau, D., Briggs, B., Dawson, T., 2004. Quantifying forest above ground carbon content using LiDAR remote sensing. Remote Sens. Environ. 93, 368–380.

Pereira, H., 2007. Cork: Biology, production and uses. Elsevier, Amsterdam 336 p.

Pignard, G., Dupouey, J., Arrouays, D., Loustau, D., 2004. Carbon stocks estimates for French forests. Biotechnol. Agron. Soc. Environ. 4, 285–289.

Sánchez-González, M., Tomé, M., Montero, G., 2005. Modelling height and diameter growth of dominant cork oak trees in Spain. Ann. For. Sci. 62, 633–643.

Sánchez González, M., del Río, M., Cañellas, I., Montero, G., 2006. Distance independent tree diameter growth model for cork oak stands. For. Ecol. Manage. 225, 262–270.

Sánchez-González, M., Cañellas, I., Montero, G., 2007. Base-age invariant cork growth model for Spanish cork oak (*Quercus suber* L.) forests. Eur. J. Forest Res. DOI 10.1007/s10342-007-0192-4.

Sprugel, D.G., 1983. Correcting for bias in log-transformed allometric equations. Ecology 64 (1), 209–210.

Tomé, M., 2004. Modelo de crescimento e produção para a gestão do montado de sobro em Portugal. Relatório final do projecto POCTI/AGR/35172/99. Publicações GIMREF RFP 1/2004. Universidade Técnica de Lisboa, Instituto Superior de Agronomia, Centro de Estudos Florestais, Lisboa.

Tomé, M., Cortiçada. A., Barreiro, S., Paulo, J.A., Meyer, A., Ramos, T., 2007. Inventário Florestal 2005–2006. Áreas, volumes e biomassas dos povoamentos florestais. Relatório resultante do protocolo de cooperação DGRF/ISA no âmbito do Inventário Florestal Nacional de 2005–2006. Publicações GIMREF RT 4/2007. Universidade Técnica de Lisboa, Instituto Superior de Agronomia, Centro de Estudos Florestais, Lisboa.

Tomé, M., Meyer, A., Ramos, T., Barreiro, S., Faias, S.P., Paulo, J., Cortiçada, A., Correia, A., 2007. Equações para a estimação de variáveis da árvore desenvolvidas no âmbito do tratamento dos dados do Inventário Florestal Nacional 2005–2006. Publicações GIMREF. RT 3/2007. Universidade Técnica de Lisboa, Instituto Superior de Agronomia, Centro de Estudos Florestais, Lisboa.

Managing Forest Ecosystems

1. K. von Gadow, T. Pukkala and M. Tomé (eds.): *Sustainable Forest Management.* 2000 ISBN 0-7923-6356-6
2. K. von Gadow (ed.): *Risk Analysis in Forest Management.* 2001 ISBN 0-7923-6900-9
3. D.L. Schmoldt, J. Kangas, G.A. Mendoza and M. Pesonen (eds.): *The Analytic Hierarchy Process in Natural Resource and Environmental Decision Making*. 2001 ISBN 0-7923-7076-7
4. K. von Gadow, J. Nagel and J. Saborowski (eds.): *Continuous Cover Forestry*. Assessment, Analysis, Scenarios. 2002 ISBN 1-4020-0659-4
5. H. Sverdrup and I. Stjernquist (eds.): *Developing Principles and Models for Sustainable Forestry in Sweden*. 2002 ISBN 1-4020-0999-2
6. T. Pukkala (ed.): *Multi-objective Forest Planning.* 2002 ISBN 1-4020-1097-4
7. G.J. Arthaud and T.M. Barrett (eds.): *Systems Analysis in Forest Resources.* Proceedings of the Eighth Symposium, held September 20-27, 2000, Snowmass Village, Colorado, USA. 2003 ISBN 1-4020-1256-X
8. R.A. Monserud, R.W. Haynes and A.C. Johnson (eds.): *Compatible Forest Management*. 2003 ISBN 1-4020-1536-4
9. C.G. Williams (ed.): *Landscapes, Genomics and Transgenic Conifers*. 2006 ISBN 1-4020-3868-2
10. A. Kangas and M. Maltamo (eds.): *Forest Inventory*. Methodology and Applications. 2006 ISBN 1-4020-4379-1
11. G. Shao and K.M. Reynolds (eds.): *Computer Applications in Sustainable Forest Management*. Including Perspectives on Collaboration and Integration. 2006 ISBN 1-4020-4305-8
12. D. Röser, A. Asikainen, K. Raulund-Rasmussen and I. Stupak (eds.): *Sustainable Use of Forest Biomass for Energy*. A Synthesis with Focus on the Baltic and Nordic Region. 2007 ISBN 978-1-4020-5053-4
13. A. Van Laar and A. Akça: *Forest Mensuration*. 2007 ISBN 978-1-4020-5990-2
14. D.M. Adams and R.W. Haynes (eds.): *Resource and Market Projections for Forest Policy Development*. Twenty-five Years of Experience with the US RPA Timber Assessment. 2007 ISBN 978-1-4020-6308-4
15. K. von Gadow and T. Pukkalo (eds.): *Designing Green Landscapes*. 2008 ISBN 978-1-4020-6758-7
16. A. Kangas, J. Kangas and M. Kurttila: *Decision Support for Forest Management*. 2008 ISBN 978-1-4020-6786-0
17. F. Bravo, V. LeMay, R. Jandl and K. von Gadow (eds.): *Managing Forest Ecosystems*. The Challenge of Climate Change. 2008 ISBN 978-1-4020-8342-6

springer.com